

Appendix A. Bibliography

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Appendix B. Estuarine and Marine Assessment Endpoints and Water Quality Indicator Variables Literature Review

To identify and evaluate biological assessment endpoints and water quality indicator variables pertinent to numeric criteria derivation, EPA searched scientific literature databases, including Google Scholar, Web of Science, and state research and agency reports. Additional information and literature were identified through workshops held by the Florida Department of Environmental Protection (FDEP). More than 800 papers and reports were reviewed to identify assessment endpoints and the likely stressors driving responses. The literature review also captured the nature of the investigations, where the studies were located, and whether models had been developed to link the assessment endpoint to nutrients or other causal variables. A detailed bibliography is provided in Appendix A.

The remainder of this appendix examines assessment endpoints (e.g., seagrass) or nutrient response indicator variables (e.g., water clarity) that could be used to derive numeric criteria. The purpose of this appendix is to demonstrate the breadth of assessment endpoints and response indicator variables that are being considered in this effort and to identify information that can provide quantifiable relationships that will inform the derivation of numeric criteria. The number of papers considered reflects the current literature review to date.

B.1 Water Clarity

Clarity is an indicator of the amount of dissolved colored material (consisting of dissolved organic matter) or suspended material in water. Natural and introduced materials affect clarity, such as suspended algae, dissolved organic matter, suspended detritus, and suspended inorganic material.

Clarity can both increase and decrease productivity. Reduced clarity increases light attenuation, making less light available to benthic algae and plants (Lee et al. 2007). Color, often a dark, tannic color resulting from freshwater discharge (Doering et al. 2006) and suspended algae, also contribute to reduced clarity and light attenuation. McPherson and Miller (1994), for example, reported that water color contributed an average of 13 percent light attenuation in Tampa Bay, 22 percent in Charlotte Harbor (overall) and 49 percent in the Upper Charlotte Harbor. For estuaries, water clarity has been a concern implicated in the decline of seagrass diversity, density, and distribution (Duarte 1991; Lee et al. 2007; Dennison 1987). In addition, it is likely that algal turbidity, such as that associated with blooms, has an impact on aesthetics and recreation (USEPA 2001). As a result, clarity is an important consideration for numeric criteria development.

In the context of numeric criteria, the relevance of clarity is related to contributions from suspended algal material in the water column. This material can come from true phytoplankton or from tychoplankton (i.e., algae dislodged or sloughed from the benthos). In either case, significant correlations have been drawn between the amount of algae in the water column, which is driven in part by nitrogen and phosphorus concentrations, and the water's clarity. In

fact, one of the common trophic indices for lakes is the Trophic State Index, which can be derived using a measure of transparency (Carlson 1977).

Clarity can be measured in a number of ways. One of the traditional measures is the vertical Secchi depth transparency measurement. Similar to clarity is turbidity, which is the measure of the scatter and absorption of light by suspended particles or, alternatively, the measure of the mass of total suspended material after filtration. Those measures are fairly convenient and, with the exception of gravimetric suspended solids, can be directly measured in the field. The best measure of light attenuation, of particular importance to seagrasses, is direct measure of photosynthetically active radiation (PAR) to generate a light extinction coefficient (k_d). Relationships have been developed to help convert between Secchi depths and k_d for specific estuaries (Dennison 1987). Equations have also been developed to parse the effects of the different contributing factors of light attenuation, and these have been applied to isolate specific effects due to nutrient-related factors (e.g., algae) (Kirk 1983; McPherson and Miller 1994; Kelble et al. 2005). These approaches have been used previously for nutrient management purposes. Specifically, they have been used in combination with light requirements for seagrasses, to set nutrient reduction loads to meet clarity targets (Steward et al. 2005; Steward and Green 2007).

A number of sources provide context for the use of clarity as a potential endpoint, as synthesized in Table B-1, including endpoints provided as part of FDEP's estuarine numeric criteria workgroup presentations. Lee et al. (2007) provide a comprehensive synthesis of light attenuation endpoints related to seagrasses, organized by seagrass species. The three relevant to Florida, *Halodule wrightii*, *Syringodium filiforme*, and *Thalassia testudinum*, have similar average light requirements of 21, 23, and 18 percent of surface irradiance, respectively. Those correspond to compensation irradiance values of 90, 76, and 60 $\mu\text{mole}/\text{m}^2/\text{s}$, which are consistent with other studies of the same taxa (Dennison 1987; Steward et al. 2005; Corbett and Hale 2006; Steward and Green 2007; Duarte 1991). EPA is considering the use of seagrass light requirements to help quantify the relationship to nitrogen/phosphorus pollution (See Chapter 2).

Table B-1. Clarity information relevant to developing numeric criteria endpoints

Characteristic	Value	Citation
Turbidity	Secchi depths High (< 1 m) Medium (>1 m, < 3 m) Low (> 3 m)	(Bricker et al. 2003b)
<i>Zostera marina</i> growth, experimental light reduction	4X reduction in light = 4X reduction in growth	(Backman and Barilotti 1976)

Characteristic	Value	Citation
<i>Z. marina</i>	<p>Saturation irradiance (I_k $\mu\text{mole}/\text{m}^2/\text{s}$) and compensation irradiance (I_c $\mu\text{mole}/\text{m}^2/\text{s}$) $I_k = 7$ $\mu\text{mole}/\text{m}^2/\text{s}$ $I_c = 1$ $\mu\text{mole}/\text{m}^2/\text{s}$</p> <p>Light compensation depth (Z_c, m) $Z_c = 0.85$ (Secchi depth)</p> <p>Light extinction coefficient (k_d, m^{-1}) $k_d = 1.7$ / Secchi depth</p> <p>Rearranging $Z_c = 1.62$ / Secchi depth is maximum depth of <i>Z. marina</i>.</p>	(Dennison 1987)
Seagrass	<p>Maximum seagrass colonization depth (Z_{col}, m) $Z_{col} = 1.87 / k_d$</p> <p>Maximum seagrass colonization depth at 10% light levels (Z_{loc}, m) $Z_{loc} = 2.30 / k_d$</p>	(Duarte 1991)
Seagrass light needs	<p>Average percent of surface irradiance needed: <i>Halodule wrightii</i>: 21% <i>Syringodium filiforme</i>: 23% <i>Thalassia testudinum</i>: 18%</p> <p>Compensation Irradiance (I_c $\mu\text{mole}/\text{m}^2/\text{s}$) <i>H. wrightii</i>: 90 <i>S. filiforme</i>: 76 <i>T. testudinum</i> : 60</p> <p>Saturation Irradiance (I_k $\mu\text{mole}/\text{m}^2/\text{s}$) <i>H. wrightii</i>: 338 <i>S. filiforme</i>: 257 <i>T. testudinum</i>: 253</p>	(Lee et al. 2007)
Depth limits	<p><i>H. wrightii</i> depth limits Median: 0.8 m–1.8 m Maximum: 1.2 m–2.8 m</p> <p><i>H. wrightii</i> annual percent surface irradiance needed: Average: 33% Minimum: 20%</p>	(Steward and Green 2007; Steward et al. 2005)

Characteristic	Value	Citation
Seagrass light targets in Charlotte Harbor	Turbidity: 5–23 NTU Chl <i>a</i> : 7-29 µg/L Percent surface light irradiance: 25% k_d : 0.63	Corbett and Hale (2006) in Martin and Martin (2006)
PRLG target for tidal Peace and Myakka rivers	No more than 10% in change from natural average annual photosynthetic compensation depth (Z_c)	(Pribble 1997)
Secchi depth	Median Secchi depth in <i>good seagrass beds</i> in Indian River Lagoon – 1.1 m	(Chang 2008)
Light attenuation coefficient (k_d , m ⁻¹)	Tampa Bay targets: Old Tampa Bay: 0.83 Hillsborough Bay: 1.58 Middle Tampa Bay: 0.83 Lower Tampa Bay: 0.63	(Greening et al. 2008)
Transparency	No decrease more than 10% over natural background	(FDEP 2008)
Seagrass light needs	Average percent of surface irradiance needed <i>S. filiforme</i> : 23%–37%	(Corbett 2008)
Waterbody light targets for seagrasses	Average percent of surface irradiance needed Tampa Bay Target: 20.5% Sarasota Bay Target: 25%–50% Charlotte Harbor Target: 25% Light extinction coefficient (k_d , m ⁻¹) Charlotte Harbor Target k_d = 0.6 Color (platinum-cobalt units [PCU]) Charlotte Harbor Target: 24 Turbidity (nephelometric turbidity units [NTU]) Charlotte Harbor Target: 24 Chlorophyll <i>a</i> (µg/L) Charlotte Harbor Target: 6.9	(Corbett 2008)

B.1.1 Scientific Justification for Not Proposing Water Clarity as a Indicator Variable

Florida currently has a water quality criterion for “transparency” which “shall not be reduced by more than 10% as compared to the natural background value.” The use of transparency or water clarity as an indicator variable is confounded by the fact that clarity can be affected by

constituents in the water not related to nitrogen/phosphorus pollution, specifically by both natural and introduced materials including suspended algae, dissolved organic matter, suspended detritus (seston), and suspended inorganic material (tripton) (Dennison et al. 1993; Fourqurean et al. 2003). Note that water clarity is an important measurement endpoint in the health of seagrass communities, as discussed further in Section C.5.

B.2 Dissolved Oxygen

Oxygen dissolves into the water either directly from the atmosphere or it is produced through photosynthesis by plants or algae living in the water, and it is routinely measured as dissolved oxygen (DO). If consumption exceeds production, DO can decline to hypoxic levels (< 2 mg/L), or anoxic levels (< 0.1 mg/L). The amount of information on the hypoxia of the northern Gulf of Mexico is growing (Rabalais 2002; Rabalais et al. 2001), along with information from Long Island Sound (e.g., Howell and Simpson 1994), Chesapeake Bay (e.g., Batiuk et al. 2009; Breitburg 1990; Hagy et al. 2004), Pamlico Sound (e.g., Stanley and Nixon 1992), and many others. Verity et al. (2006) observed trends of decreasing DO in well-mixed Georgia estuaries concomitant with increased nutrient concentrations, chlorophyll, and bacterial respiration over a 20-year period. There is evidence that increased nutrient loading has been contributing to occurrences of decreased DO, or hypoxia, in Florida Bay and the Florida Keys (Lapointe and Clark 1992). Many other estuaries along the Gulf of Mexico experience periodic hypoxia: EPA's EMAP survey sampled DO along the entire U.S. portion of the Gulf of Mexico coast from the Rio Grande Estuary in Texas to Tampa Bay, Florida (1991–1994) and estimated that 5.2 percent of the total estuarine surface area had DO below 2.0 mg/L (Engle et al. 1999).

B.2.1 Effects

The cause and effect relationship between nitrogen/phosphorus pollution and marine and estuarine hypoxia is clear and unequivocal on a global scale (Conley et al. 2009a; Conley et al. 2009b; Conley et al. 2009c; Diaz 2001; Diaz and Rosenberg 2008). Table B-2 summarizes literature findings on trends, causes, and effects of hypoxia in estuaries and coastal areas. A large hypoxic zone in an estuary can lead to substantial changes in fish, benthic, and plankton communities. The lack of oxygen forces fish and mobile benthic invertebrates to migrate out of an area and represents a habitat loss. In extreme cases, anoxia can lead to fish kills (e.g., Howell and Simpson 1994; Kidwell et al. 2009). Benthic organisms that cannot escape are variably affected as the oxygen levels decline. Chronic hypoxia, and short-term but recurring anoxia, cause marked changes in the benthic invertebrate community of estuaries (e.g., Baker and Mann 1992; Baker and Mann 1994b; Baker and Mann 1994a; Baustian and Rabalais 2009; Breitburg 2002). In addition to mortality from chronic hypoxia, motile organisms might be able to leave the hypoxic zone, sometimes resulting in jubilees (i.e., massive concentrations of crabs and fish escaping from a hypoxic tidal wedge or seiche) (e.g., Diaz 2001). Finally, even intermittent hypoxia can cause benthic assemblage changes to resistant or tolerant organisms and reduces fish predation pressure in the hypoxic zone, because the fish avoid it (e.g., Kidwell et al. 2009). The habitat exclusion means that smaller individuals, especially juveniles, might be more exposed and more susceptible to predation (e.g., Breitburg 2002). Hypoxia has been implicated in the recent increases and late-summer dominance of gelatinous zooplankton (jellyfish and ctenophores) in Chesapeake Bay and other eastern estuaries, because the gelatinous zooplankton

tend to be more tolerant to hypoxia (Grove and Breitburg 2005). If the hypoxia extends into shallow waters, it could affect spawning and nursery areas of important fish species.

The effects of hypoxia in coastal regions are generally the same as in more enclosed estuaries, with one exception: since highly mobile pelagic fish have a larger area in which to escape, population effects of hypoxia on pelagic fish have not been documented (Switzer et al. 2006). Flatfish and other benthic feeding fish are excluded from the Gulf of Mexico hypoxic zone during summer and early fall, potentially reducing their foraging ability and populations because of habitat loss (Switzer et al. 2009). Outside the Gulf hypoxic zone, fishery production has remained stable or increased because of the fertilization effect of the nutrient pollution (Switzer et al. 2009).

B.2.2 Mechanisms

DO is produced by photosynthesis in the photic zone of the water column, and it also enters the water column by diffusion from the atmosphere at the surface and advection (circulation) of oxygen-rich surface waters to deeper areas. Circulation is increased by wind and tide, and it is decreased by calm conditions and stratification, where the surface layer of water is lighter than the deep water. In estuaries, salinity gradients are the most common cause of stratification (e.g., Diaz 2001; Hagy et al. 2004), where lighter, fresh water flows in on top of denser and more saline water. Temperature can also be another contributing factor to stratification (e.g., Stanley and Nixon 1992), where warm, lighter surface water (heated by sunlight) overlies denser, cooler water. DO can fluctuate widely within hours, owing to wind-induced mixing, tides, wind-induced seiches, and day and night. Tides and seiches can move low-DO bottom water into nearshore zones (e.g., Breitburg 1990); daytime photosynthesis increases DO and nighttime respiration decreases DO (e.g., Breitburg 1990), and onset of wind can mix an unstratified but stagnant body of water.

The pathway of eutrophication leading to hypoxia is well known: increases in the concentrations of N or P can trigger excess algal growth. The organic algal biomass then decomposes, which consumes oxygen, depleting the water column of DO. Organic loading by itself (such as raw sewage or untreated pulp mill effluent) can also cause hypoxia. Although oxygen is produced by algal growth, both respiration by the algae and decomposition (bacterial respiration) and respiration by other organisms deplete the oxygen. The combined respiration rates can use up the oxygen at night and in deep waters where there is insufficient light to support photosynthesis.

Hypoxia and anoxia in bottom waters result in anoxia in the surface sediments, sometimes setting up severe reducing zones. The reducing environment of the sediment has geochemical consequences, including release of toxic hydrogen sulfide (H_2S), SRP, and ammonia (NH_3) (Diaz and Rosenberg 2008; Kemp et al. 2005; McCarthy et al. 2008). The sediment of hypoxic zones, thus, becomes a potential source of nutrients that can increase the degree of eutrophication.

Systems that have had persistent and chronic hypoxia often fail to recover even after pollution loadings have been reduced (e.g., Conley et al. 2009a; Conley et al. 2007; Diaz and Rosenberg 2008), possibly because of the sediment having become a nutrient source. Hypoxia creates a carbon/energy sink in the hypoxic deep water: the carbon is removed from the food chain and is

not available to top predators (many commercial and sport fish) or filter feeders (oysters, clams). Reduced fishery production of hypoxic zones has been documented worldwide (e.g., Diaz and Rosenberg 2008), although it is offset to some extent by increased fisheries production at the margins of the hypoxic zone.

Table B-2. Summary of literature findings on trends, causes, and effects of hypoxia in estuaries and coastal areas.

Response	Cause/ Predictor	Location Estuary (E) or Coastal (C)	Equation Describing Relationship?	Synopsis	Reference
Chlorophyll, DO	Nutrients	Neuse River (E)	No	Long-term trend monitoring of nutrients and response	(Burkholder et al. 2006)
DO, hypoxia	diurnal, physical	continental shelf of Louisiana (C)	No	early observation of Gulf hypoxia	(Rabalais et al. 1994)
DO, hypoxia	discharges and their nutrient flux, sediment records	inner- to mid- Louisiana continental shelf (C)	No	relation of nutrient pattern and sediment records with DO levels	(Rabalais et al. 2001)
DO, hypoxia	sediment paleoindicators	Gulf of Mexico (C)	No	sediment indicators used to describe history of hypoxia and eutrophication. no hypoxia before 1900. hypoxia increased after 1950 to present-day max. Phytoplankton production increased in 20th C, sharply increased after 1950. Ag increase 1900-1950, but N fertilizer started 1950.	(Rabalais et al. 2007)
DO, hypoxia	N loading	Chesapeake Bay (E)	Yes	A Simple Model for Forecasting the Effects of Nitrogen Loads on Chesapeake Bay Hypoxia. Target 35% N loading reduction confirmed.	(Scavia et al. 2006)
Eutrophication	sediment indicators			Extent in time. See Kemp et al. (year?) for comprehensive reconstruction	(Cornwell et al. 1996)
Eutrophication	sediment records	Chesapeake Bay (E)	No	reconstruction of the progression of eutrophication and anoxia/hypoxia over the past five centuries	(Zimmerman and Canuel 2002)
Hypoxia		(C, E)	No	review of hypoxia and causes	(Diaz 2001)

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Response	Cause/ Predictor	Location Estuary (E) or Coastal (C)	Equation Describing Relationship?	Synopsis	Reference
Hypoxia	N, P, discharge	Gulf of Mexico (C)	Yes	relationships between the area of bottom water hypoxia along the northern Gulf of Mexico and Mississippi–Atchafalaya River nitrate concentration, total phosphorus (TP) concentration, and discharge	(Greene et al. 2009)
Hypoxia	N	Chesapeake Bay (E)	Yes	long-term pattern of hypoxia and anoxia in Chesapeake Bay and its relationship to NO _x -loading. Requires 40% reduction of N loading	(Hagy et al. 2004)
Hypoxia	nitrogen, among others	Gulf of Mexico (C)	No	periphyton of net production, respiration, nutrient content, and biomass. Requires 40-45% reduction in N loading	(Justic et al. 2007)
Hypoxia	benthic forams	Chesapeake Bay (E)	No	benthic foraminifera (protists) were used as bioindicators to estimate the timing and degree of changes in dissolved oxygen over the past five centuries. Low DO correl with N loading, also mod by river flow	(Karlsen et al. 2000)
Hypoxia	N	Gulf of Mexico (C)	No	comparison of three models of N and hypoxia. large-scale hypoxia in 1970s; 30% N load reduction not sufficient.	(Scavia et al. 2004)
Hypoxia, benthos, fish	nutrients	Gulf of Mexico (C)	No	hypoxia and its record in sediment and effects on benthos and sediment chemistry	(Rabalais et al. 2002)
Hypoxia, DO	N, P, silicate loading	Charlotte Harbor, FL (E)	Yes	influence of nutrient loading on summertime low oxygen zone. N loading causal agent for hypoxia; 3 X higher than 1800 baseline	(Turner et al. 2006a)
Phytoplankton, clarity, DO, and SAV	nutrient loading	Choptank and Patuxent Rivers (E)	No	Nutrients and response over time	(Fisher et al. 2006)

Response	Cause/ Predictor	Location Estuary (E) or Coastal (C)	Equation Describing Relationship?	Synopsis	Reference
Phytoplankton, DO	sediment indicators	northern Gulf of Mexico (C)	No	reconstructed water quality changes 1800 to 2000. TN drives hypoxia	(Turner et al. 2006b)
Various endpoints		(E)	No	Review and history of Bay	(Kemp et al. 2005)
Various endpoints	nutrients	(C)	No	review of two complementary approaches to assessing the causes and consequences of eutrophication	(Scavia and Bricker 2006)

B.2.3 Paleoecological Investigations

Paleoecological investigations (benthic foraminifera in cores) have shown that hypoxia did not occur (or was exceedingly rare) prior to 1900 in the northern Gulf of Mexico, in contrast to suggestions that hypoxia in the Mississippi River plume was natural (Rabalais et al. 2007). Hypoxia occurred occasionally up to around 1950, and after 1950 became more frequent and widespread (Rabalais et al. 2007). Coinciding with the hypoxia, algal pigments in the sediments below the hypoxic zone increased during the same period, indicating large increases in algal production and biomass in the Mississippi plume in the years after 1950 (Rabalais et al. 2007).

Paleoecological investigations in Charlotte Harbor demonstrated increases in phytoplankton and in geochemical indicators of hypoxia during the 20th century (Turner et al. 2006a). N loading was estimated to have increased by more than three times over a baseline, natural condition estimated for 1800. Although freshwater inflow has decreased in Charlotte Harbor (increasing net residence time of the water, making the estuary more susceptible to hypoxia), weight of evidence was strong that N loading is the most important causal agent for the phytoplankton growth and hypoxia (Turner et al. 2006a).

B.2.4 Potential Measurement Endpoints

State and federal agencies monitor indicators of eutrophication in estuaries including concentrations of N and P, chlorophyll *a*, reports of algal blooms, and DO. Florida's current numeric criterion for DO specifies that DO "shall not average less than 5.0 in a 24-hour period and shall never be less than 4.0. Normal daily and seasonal fluctuations above these levels shall be maintained."

B.3 Phytoplankton

Phytoplankton are microscopic, free-floating autotrophic organisms that inhabit aquatic ecosystems. While the water column also contains imported riverine phytoplankton and tychoplankton, especially during and after periods of turbulent flow, the phytoplankton represents an important component of estuarine food webs through primary production. While

the phytoplankton can include mixotrophic species (taxa that acquire energy through autotrophic and heterotrophic pathways, typified by some dinoflagellates), it is typically dominated by autotrophic taxa. Chlorophyll *a* is used as a surrogate measure of phytoplankton abundance (or biomass) in the water column, as well as serving as an index of the productivity and trophic condition of waters. Chlorophyll *a* biomass reflects the *standing stock*, which is the balance of growth and loss in pelagic waters. The benefits of chlorophyll *a* as an indicator are its relevancy to conditions of the Florida Bay ecosystem, its sensitivity to stressors such as nutrients, and ease of monitoring (Boyer et al. 2009). Higher concentrations of chlorophyll *a* are indicative of overproduction of algae, which may be related to nitrogen/phosphorus pollution. A weakness of chlorophyll *a* as a measure of phytoplankton is the variability of cellular chlorophyll content among species (Boyer et al. 2009).

B.3.1 Components of Phytoplankton

Phytoplankton can be characterized by biomass and composition. Biomass refers to the mass of phytoplankton in the water column (i.e., the amount present), and composition refers to the taxonomic makeup of the phytoplankton assemblage (i.e., the diversity of species present). Biomass is typically measured using extraction and analysis of photopigments with a variety of methods, the most common being the spectrophotometric or fluorescence analysis of chlorophyll *a*. Photopigment-based analyses are an estimate of biomass only, as photopigment composition varies across different algal taxa and even within taxa across a variety of environmental conditions. Biovolume estimation based on microscopy or cytometry and imaging is another approach for biomass estimation, as well as gravimetry to estimate dry and ash-free dry mass. Composition is primarily measured with collection and microscopic identification to the lowest practical taxonomic level, and other methods are evolving as gene-based techniques improve. Chromatographic separation of photopigments has allowed biomass and gross assemblage composition analysis to be analyzed simultaneously and is being used more often (Mackey et al. 1996; Pinckney et al. 2001; Millie et al. 2004; Paerl et al. 2006; Paerl et al. 2007).

B.3.2 Effects of Nitrogen/Phosphorus Pollution on Phytoplankton

Both light and nitrogen/phosphorus supply are major drivers and principal limiting factors of phytoplanktonic growth. Light is controlled mainly by three factors: incident irradiance, attenuation of PAR, and mixing depth (Bledsoe and Phlips 2000). Phytoplanktonic species vary in their requirement and competitive ability for nutrients and light. Other factors that increase or decrease phytoplankton biomass include flow/residence time, consumption/grazing, temperature, and salinity. For example, McPherson and Miller (1994) found that chlorophyll *a* caused light attenuation of 16-28 percent across Charlotte Harbor and Tampa Bay.

The capacity for nutrients to cause increased primary productivity and biomass is undisputed, and increases in chlorophyll with nutrient concentration have been observed in numerous estuarine observational and experimental studies, as shown in Table B-3 (e.g., Vitousek et al. 1997; Fisher et al. 2006). Algal growth depends on a variety of factors that vary in both space and time that can make characterizing the relationships between nutrient loads or concentrations and algal biomass problematic (Murrell et al. 2007; Hagy et al. 2008). Both nitrogen (N) and phosphorus (P) limit algal growth in Florida estuaries, but nutrient limitation can vary by estuary or even within estuaries (Fourqurean et al. 1993; Hoyer et al. 2002; Phlips et al. 2002; Murrell et

al. 2002; Phlips et al. 2004a; Juhl and Murrell 2005; Phlips et al. 2004b). Variation in the nutrient-chlorophyll relationship has been attributed to a variety of factors including residence time (Boynton et al. 1982; Kennish 1986; Knoppers et al. 1991; Monbet 1992; Phlips et al. 2004b; Verity et al. 1998) and grazing (Cloern 1982; Officer et al. 1982), among others.

The effect of nitrogen/phosphorus pollution on phytoplanktonic assemblage composition such as nuisance algal blooms, macroalgal increases, reduced desired phytoplankton is well-established (Hutchinson 1961) and has been observed along nutrient gradients in Florida and elsewhere (e.g., Piehler et al. 2004; Sanders et al. 1987; Parsons et al. 1978; Paerl 1988; Harding 1994; Richardson 1997; Hagy et al. 2008).

Table B-3. Phytoplankton information relevant to developing numeric criteria endpoints

Characteristic	Response	Citation
Chlorophyll <i>a</i>	Hypereutrophic: > 60 µg/L Eutrophic: 20-60 µg/L Mesotrophic: 5-20 µg/L Oligotrophic: <5 µg/L	(Bricker et al. 2003b)
Chlorophyll <i>a</i>	< 5 µg/L = unimpacted	(Nixon and Pilson 1983)
Chlorophyll <i>a</i>	> 5 µg/L is dangerous for corals	(Lapointe and Matzie 1996)
Chlorophyll <i>a</i>	> 20 µg/L, SAV declines and seagrass community shifts to monoculture	(Stevenson et al. 1993; Twilley et al. 1985)
Chlorophyll <i>a</i>	> 60 µg/L, High turbidity and low bottom water DO	(Jaworski 1981)
Chlorophyll <i>a</i>	Chlorophyll correlated with total N load and DO changes in Chesapeake Bay.	(Boynton et al. 1996)
Chlorophyll <i>a</i>	Relevant, sensitive, feasible, defensible, and integrative indicator for Florida Bay: Using EPA approach, 75 th percentiles: 0.43 to 4.9 µg/L across S. Florida estuaries	(Boyer et al. 2009)
Chlorophyll <i>a</i>	Thresholds for Denmark (depending on coastal waterbody class): Reference: 1-6 µg/L High to Good: 1-8 µg/L Good to Moderate: 1-10 µg/L Moderate to Poor: 2-12 µg/L Poor to Bad: 2-13 µg/L	(Carstensen and Henriksen 2009)
Chlorophyll <i>a</i>	Log ₁₀ Chlorophyll <i>a</i> = -1.13 + 1.17log ₁₀ TP (r ² = 0.81) Log ₁₀ Secchi depth = 0.48 + - 0.48log ₁₀ Chlorophyll <i>a</i> (r ² = 0.68) (300 near shore samples along entire Fl Coast)	(Hoyer et al. 2002)
	<u>Mean</u> <u>Min</u> <u>Max</u>	

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Characteristic	Response	Citation
	Chl <i>a</i> 3.7 0.2 28.2 TP (µg/L) 25 3 122 TN (µg/L) 309 93 1337	
Chlorophyll <i>a</i>	Nutrients predict chlorophyll <i>a</i> in Caloosahatchee, especially the upper estuary and chlorophyll <i>a</i> explains 63% of variation in DO concentration 2 months later; Affected by water residence time	Doering et al. (2006) in Martin and Martin (2006)
Chlorophyll <i>a</i>	Response of chlorophyll <i>a</i> to DIN differs in estuaries with > 2m tidal range from < 2m range because of increased turbidity, flushing, etc	(Monbet 1992)
Chlorophyll <i>a</i>	Potential WQ target for PLRG in Tidal Peace and Myakka: Maximum monthly chlorophyll <i>a</i> : < 60 µg/L (hypereutrophic)	(Pribble 1997)
Chlorophyll <i>a</i>	Decreased from 10 to 4.5 µg/L in Boston Harbor from 1995 to 2005 with: TP decrease from 2.25 to 1.75 µM TN decrease from 33 to 20 µM DIN decrease from 13 to 5 µM	(Taylor 2005)
Chlorophyll <i>a</i>	Chlorophyll <i>a</i> decreased in upper estuary increased in lower estuary of Patuxent River Estuary, Chesapeake Bay in response to: 40-60% decrease in point-source DIN and DP loading TN decrease from 125-250 µM to <125 µM TP decrease from 4-12 µM to <4 µM (until recently)	(Testa et al. 2008)
Chlorophyll <i>a</i>	Tampa Bay chlorophyll <i>a</i> targets: Old Tampa Bay: 8.5 µg/L Hillsborough Bay: 13.2 µg/L Middel Tampa Bay: 7.4 µg/L Lower Tampa Bay: 4.6 µg/L	(Greening et al. 2008)
Chlorophyll <i>a</i>	FDEP IWR goal: 11 µg/L Charlotte Harbor Goal: 6.9 µg/L	(Corbett 2008)
Chlorophyll <i>a</i>	Concentration associated with healthy seagrass beds in Indian River Lagoon: 4 µg/L	(Chang 2008)
Composition	Diatoms dominate areas of Indian River Lagoon with intermediate turnover and high P loading. Common bloom forming taxa: <i>Skeletonema costatum</i> , <i>Dactyliosolen fragilissimus</i> , <i>S. menzelii</i> , <i>Cerataulina pelagica</i> , <i>Odontella regia</i> , <i>Chaetoceros lorenzianus</i> , <i>Rhizosolenia setigera</i> , <i>Thalassionema nitzschoides</i>	(Badylak and Phlips 2004)
Composition	High nutrient pulse alter assemblage composition, reduce diversity, and correlated with increased probability of harmful algae	(Spatharis et al. 2007)

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Characteristic	Response	Citation
Composition	In Santa Rosa Sound, phytoplankton growth under natural conditions is controlled by grazing, but exceeds grazing control with increasing chlorophyll <i>a</i> from enrichment. Diatoms dominate after enrichment (16 µM N and 2 µM P)	(Juhl and Murrell 2005)
Composition	Algal phytoplankton species indicative of enrichment off Yucatan: <i>Heterocapsa circularisquama</i> , <i>Pyrodinium bahamense</i> , <i>Trichodesmium erythraeum</i> , <i>Nitzschia longissima</i> , etc.	(Álvarez-Góngora and Herrera-Silveira 2006)
Composition	<i>Schizothrix calcicola</i> considered an indicator of nutrient enrichment in Pensacola Bay Cyanobacteria track chlorophyll <i>a</i> in Tampa Bay	(Dixon et al. 2009)
Composition	Reliable eutrophication indicators in Finland: Oscillatoriacean cyanobacteria and diatoms <i>Cyclotella choctawhatcheeana</i> , <i>Cylindrica closterium</i> , and <i>Skeletonema costatum</i>	(Jaanus et al. 2009)
Composition	Diatoms, chlorophytes, and cryptophytes dominate during increased nutrient loading	(Arhonditsis et al. 2007)
Composition	Increased summer cyanobacteria abundance and biomass in impaired waters because of decreased light and increased nutrients in Chesapeake Bay	(Marshall et al. 2006)
Composition	Phytoplankton Index of Biotic Integrity developed for Chesapeake Bay; -Based on chlorophyll, composition, and carbon content -Discriminates degraded and reference sites	(Lacouture et al. 2006)
Composition	Pulp Mill effluent nutrients increase N and P and caused blooms of diatoms, raphidophytes, dinoflagellates, and cyanobacteria (during drought) in Perdido Bay; Both chlorophyll and composition respond to enrichment	(Livingston 2007)
Composition	Relative composition of diatoms increases with N enrichment in the summer, fall and winter; Cyanobacteria increase in spring Enrichment: 20 µM NO ₃ -N and 5 µM PO ₄ -P	(Piehler et al. 2004)
Nutrient concentrations	Limiting thresholds for phytoplankton in Chesapeake Bay: 0.07 mg DIN-N/L 0.007 mg PO ₄ -P/L	(Fisher et al. 2003)
Nutrient concentrations	Tidal fresh, oligo- and polyhaline: 0.003 mg PO ₄ -P/L Mesohaline: 0.002 mg PO ₄ -P/L	(USEPA 2003)
Nutrient concentrations	Algal production increases from 0 to 2 mol N/m ² /yr	(Cloern 1999)
Nutrients	Increase in P of 1.5-10x Increase in N of 1.4-2.6x	(Bundy et al. 2003)

Characteristic	Response	Citation
	Resulted in nearly 7-fold increased in Chlorophyll <i>a</i> in experimental mesocosms	

B.3.3 Effect of Phytoplankton on Estuarine and Coastal Waters

While phytoplankton is an important source of energy to the food web that supports secondary production, excess primary production can cause a variety of negative effects (Vitousek et al. 1997; Bricker et al. 2003b). Excess growth can reduce water clarity, resulting in reduced light availability for benthic algae, macrophytes, and seagrasses (Bricker et al. 2008; Boyer et al. 2009). In addition, seagrass decomposition and destabilization of sediments can result in more nutrient inputs into the water column from sediments (Boyer et al. 2009). Excess primary production also provides greater loads of reduced carbon, which fuels respiration, decreases DO, and results in hypoxic and anoxic conditions (Vitousek et al. 1997). Such conditions are harmful to many invertebrate and vertebrate species and can lead to reductions in their diversity, distribution, and secondary production. Reduced oxygen also affects a variety of other biogeochemical processes that can affect water quality, including metal speciation. Lastly, excess algal growth can affect recreational use through perceived effects on aesthetics and real threats to well-being vis-à-vis nuisance and toxic algal species (Paerl 1988).

B.3.4 Potential Assessment Endpoints

EPA is considering two assessment endpoints based on phytoplankton: (1) balanced phytoplankton biomass and production, and (2) balanced benthic and planktonic biological assemblages (see Chapter 2).

B.4 Macroalgae

Macroalgae are large aquatic plants that can be seen by the naked eye and occur in many colors and forms, such as, green, red, brown, blue, tall and mat-like. Macroalgae are eukaryotic, photosynthesizing, multi-celled organisms (Schmidt 2003a). Green (Chlorophyta), Red (Rhodophyta) and Brown-Kelp (Phaeophyta) are the most common groups (Hauter and Hauter 2010). Green algae, whose green coloring comes from chlorophyll, are usually unicellular, but some species are considered multicellular/colonial. They are usually found in freshwater but also inhabit marine bays and estuaries (Schmidt 2003b). Red algae contain red pigment called phycobilins and inhabit shallow marine water. They also help to create coral reefs from calcium carbonate (Schmidt 2003d). Brown algae have pigments like fucoxanthin and are most common near rocky coastlines. The most common brown algae are kelp, which grow very large and form offshore masses, also known as kelp forests (Schmidt 2003c).

Hanisak and Blair (1988) conducted a year-long study to document macroalgal seasonal and depth distribution on the eastern continental shelf of Florida, finding 208 taxa from the area, of which 42 (20.2 percent), 19 (9.1 percent), and 147 (70.7 percent) belonged to the Chlorophyta, Phaeophyta, and Rhodophyta taxa, respectively (Hanisak and Blair 1988).

B.4.1 Macroalgal Blooms

Macroalgal blooms are produced by nutrient enrichment in shallow estuary waters and often disturb seagrass and coral reef ecosystems (Valiela et al. 1997). They are predominantly found in inshore coastal waters and are among the first organisms to come into contact with nutrients (ECOHAB 1995). Coral reefs are known to be a common location for macroalgal blooms because of the availability of light and warm water. Macroalgal blooms do not have toxic impacts, but they do share some characteristics with toxic phytoplankton species and oftentimes are grouped with *harmful algal blooms* (HABs) (ECOHAB 1995). They differ from microalgal blooms because of a broader range of ecological effects and a tendency to last longer, for years to decades (Valiela et al. 1997). Many macroalgal blooms considered to be *nuisances* are green algae, or Chlorophyta, which accumulate in thick, unattached mats (i.e., *Ulva*, *Cladophora*, *Chaetomorpha*, *Gracilaria*) (McGlathery 2001). In nutrient-rich areas, macroalgal populations can reach a biomass of over 0.5 kilogram per square meter (kg/m²). Large blooms have also been proven detrimental to tourist attractions along shorelines (Lapointe and Bedford 2007).

Anthropogenic activities that result in nutrient discharges may add to the growth of macroalgal blooms and eutrophication (Lapointe 1997). Even small increases in N or P can cause a rapid rate of algal growth (NRC 2000). Macroalgal blooms can outgrow or kill seagrasses (Taylor et al. 1995; Duarte 1995; McGlathery 2001) by altering the competitive balance from slow-growing to fast-growing primary producers (Collado-Vides et al. 2007). The blooms can outgrow or kill corals by limiting light and inhibiting recruitment of juvenile corals (NRC 2000; Lapointe et al. 1997). Macroalgal blooms can accumulate up to 2 m on coral reef surfaces, smothering sponges and hard and soft corals (Lapointe 2007a). Also, they can cause *phase shifts* from hermatypic corals to macroalgal dominance at coral reefs (even with abundant herbivores) (Littler and Littler 1984). Finally, they can trigger coral disease (white plague type II) through physical contact with the macroalga *Halimeda opuntia*, which has resulted in mortality in Caribbean corals (Nugues et al. 2004).

Macroalgal blooms can attract grazing organisms (McGlathery 1995) and can alter community structure directly by changing competition among algal species for nutrients (Howarth et al. 2000). Macroalgal blooms can destroy the habitats of indigenous species, where the macroalgae use up oxygen and alter biogeochemical cycles (ECOHAB 1995). This leads to hypoxia/anoxia, resulting in decreased biological diversity (NRC 2000; Lapointe 1997).

Most macroalgal bloom studies were undertaken in relation to coral reefs. Below are some documented macroalgal blooms that have been observed throughout the world, and can be used to inform of the potential for similar events in Florida.

- Peel-Harvey estuary in Australia: Blooms have been present since the late 1960s. *Cladophora* was most common in deep water until 1979 and was then replaced by *Chaetomorpha*. Present now are *Ulva*, *Enteromorpha* and *Chaetomorpha*, depending on nutrient levels (Lavery et al. 1991).

- Coast of Bermuda: *Cladophora* blooms have grown since the late 1960s. Nutrient seepage from groundwater has allowed *Cladophora* to grow in mats 5–100cm thick over hundreds of acres of water (Lapointe and O'Connell 1989).
- Great Barrier Reef, Australia: Macroalgal blooms have become more common since the 1970s. They have hindered coral growth rates and have displaced the coral in some places (Bell 1992).
- Kanehoe Bay, Hawaii: Significant increases in *Dictyosphaeria* because of nutrient inputs from sewage, causing much of the coral in Kanehoe Bay to die and be replaced with the macroalgae (Smith et al. 1981).
- Coos Bay estuary, Oregon: Studied in 1981 and 1982 because of *Enteromorpha* blooms that formed macroalgal mats, leading to plant and ecosystem loss (Pregnall and Rudy 1985).
- Baltic Sea: *Cladophora* and *Enteromorpha* replaced the macrophyte, *Fucus*, hindering the growth of much of the other fauna in the ecosystem in the early 1980s (Baden et al. 1990).
- Northern Mediterranean coast: *Caulerpa taxifolia* blooms reported in the 1980s, occurring at depths of 0 to 50 m, affected local seaweed and seagrass populations (Meinesz et al. 1993).
- Nahant Bay, Massachusetts: Blooms of *Pilayella littoralis* were observed, resulting from high ammonia levels in the 1980s (Pregnall and Miller 1988).
- Venice Lagoon, Italy: Macroalgae blooms of *Ulva* were observed in the early 1990s (Sfriso et al. 1992).
- Waquoit Bay, Massachusetts: Following an increase in nutrients in the late 1980s and early 1990s, *Cladophora* and *Gracilaria* blooms formed, replacing *Zostera* (Valiela et al. 1992; Peckol et al. 1994).

The waters of Florida Bay and the Florida Reef Tract are P-rich and experience N and P discharges from agricultural and wastewater activities on land. That has caused red algae, *Laurencia intricata* and *Spyridia filamentosa*, brown algae, *Dictyota* sp. and *Sargassum filipendula* and green algae, *Enteromorpha* sp., *Codium isthmocladum*, and *Halimeda* sp. to grow in the Florida Bay area (ECOHAB 1995). That growth has hindered the commercial fish, lobster, and shrimp nurseries in the area and has caused dramatic damage to the Florida Reef (ECOHAB 1995). For example, in Florida Bay, extensive populations of *Laurencia* spp. (Rhodophyta) have been documented in the 1980s (Lapointe et al. 1994). For coral reefs in southeast Florida, an increasing number of invasive chlorophyte blooms, especially of the non-native *Caulerpa brachypus* f. *parvifolia* (Lapointe and Bedford 2010), have been overgrowing sponges, hard corals, and octocorals (Lapointe et al. 2005a). In Palm Beach and Broward counties, macroalgal blooms of unattached *Codium isthmocladum* have been historically documented to occur at low-standing crops in deep-reef ecosystems (24–58 m) in the 1970s (Hanisak and Blair 1988), while the blooms occurred in deep reefs (> 30 m) and up to 2 m thick over the reef surface (Lapointe et al. 2005c). Spectacular blooms of unattached *Codium isthmocladum* have been documented in southern Palm Beach County and northern Broward County beginning in the summer of 1990

(Lapointe et al. 2005c). Also, *Codium isthmocladum* (Chlorophyta) was found on reefs off of southern Palm Beach County (summers of 1994–1996) (Lapointe et al. 1997; Lapointe et al. 2005a). Mats up to 2 m thick (6.5 ft.) occurred in 24–40 m of water off southern Palm Beach and northern Broward counties (Jacoby et al. No date). *Caulerpa verticillata* were discovered in fall of 1997 at Singer Island just north of the *Princess Anne* artificial reef (Jacoby et al. No date). Jacoby et al. (No date) noted that *Caulerpa brachypus* was found in May 2001 at a depth of 43 m off Singer Island, as well as being widespread in waters 20–43 m deep off northern Palm Beach County; it was also found in March 2003 at Pepper Park in Fort Pierce, Indian River Lagoon around Jensen Beach causeway, and at Juniper Inlet. Jacoby et al. (No date) also reported increases of *Caulerpa racemosa* in the 1990s.

B.4.2 Factors that Control Macroalgal Growth

Studies have shown that light intensity (irradiance), temperature, water depth, presence of grazers, water movement, and desiccation can affect macroalgal growth (Lapointe 1987; Mann 1973; Nielsen et al. 2002; Dring 1981; Valiela et al. 1997). Lirman and Biber (2000) found an increasing trend in macroalgal biomass and percent cover on reefs when both light and temperature conditions are favorable in the Northern Florida Reef Tract. They found maximum percent cover in July (56.7 percent) and minimum percent cover in December (25.8 percent). In addition, they found that different algal groups dominate at different times of the year.

Light and nutrients are critical factors that control macroalgal growth and productivity. When light is plentiful, nutrients become the limiting factor (Lapointe and Tenore 1981), and when nutrients are present in abundance, irradiance plays a more important role in determining productivity (Lapointe and O'Connell 1989). Light is usually limiting during the winter and early spring, while nutrients limit growth during the summer. However, macroalgae can undergo biological changes to maximize photosynthesis and growth by obtaining nutrients from the water column or sediments (Krause-Jensen et al. 1996). Macroalgae have the ability to optimize productivity under different irradiance and nutrient limitations (both N- and P-) (Lapointe 1997). A study conducted by Lapointe and Tenore (1981) examined the effects of N additions and different light conditions on the macroalga *Ulva fasciata* (see Table B-4). They found that N additions enhance more growth in high light, and chlorophyll content increases more under increased N loading under both high and low light (Lapointe and Tenore 1981). Macroalgal blooms are known to terminate when phytoplankton prevents light from reaching the macroalgae, a condition that could be more common in areas with longer water residence times (Valiela et al. 1997).

Table B-4. Biochemical composition and growth measurements of *Ulva fasciata* cultured under different light and N loading

Treatments ^a	%N ^b	%C ^b	C:N ^b	%Ash ^{b,c}	Chlorophyll <i>a</i> (mg g/dry wt)	Carbon: Chlorophyll <i>a</i>
Initial	4.16±0.12	31.3±0.5	7.5±0.1	25	6.5±0.4	48±0
49–1.0	3.93±0.04	33.4±1.1	8.5±0.2	26	5.8±0.2	58±4
49–5.5	4.32±0.09	33.6±0.4	7.8±0.1	26	6.5±0.2	52±3
49–10.1	4.44±0.13	33.9±0.4	7.7±0.2	25	6.9±0.3	49±2
137–1.0	3.40±0.02	35.5±0.4	10.4±0.2	23	2.9±0.3	124±12
137–5.5	4.10±0.00	35.7±0.3	8.7±0.1	23	4.7±0.4	77±7

Treatments ^a	%N ^b	%C ^b	C:N ^b	%Ash ^{b,c}	Chlorophyll <i>a</i> (mg g/dry wt)	Carbon: Chlorophyll <i>a</i>
137–10.1	4.47±0.03	35.2±0.2	7.9±0.1	24	6.4±0.4	55±5
255–1.0	2.99±0.03	35.5±0.7	11.9±0.0	22	3.0±0.1	116±2
255–5.5	3.54±0.22	35.8±1.4	10.1±0.2	22	3.4±0.1	107±4
255–10.1	4.65±0.22	37.3±0.1	8.0±0.4	21	5.7±0.1	65±0

Source: Lapointe and Tenore (1981)

a. Light (ly/day) – N loading (mmol/day)

b. Dry weight basis

c. SD < 0.8

Overfishing of herbivorous fish or key grazers has been blamed for causing increased macroalgal blooms on coral reefs, but that has not been proven despite various grazer reduction experiments (Lapointe et al. 2005c). It has been debated whether *top-down* control mechanisms, such as herbivory, are the primary control mechanism that promotes excessive algal growth or whether it is affected by *bottom-up* controls such as nutrient inputs (Hughes et al. 1999). Valiela et al. (1997) points out that grazing pressure can control macroalgal growth but only under lower N conditions. Lapointe (1997) found that the abundance of herbivores is not the only control mechanism and that bottom-up mechanisms (nutrient inputs) also play a critical role. Lirman and Biber (2000) conducted an evaluation of macroalgal communities in the Florida Reef Tract, finding that there was a “general lack of correlation” between algal biomass/percent cover and the abundance and consumption rates of grazers.

Many coastal systems are often N-limited, but increasing evidence is surfacing that P limitation also exists in near-shore habitats (Howarth 1988). Evidence of both nitrogen and phosphorus limitation eliciting macroalgal responses to nutrient enrichment are highly species-specific and vary among regions (Armitage et al. 2005). Algal species differ widely in requirements and tolerances to different levels of nutrients (Howarth et al. 2000).

For example, Lapointe (1997) found that *Chaetomorpha linum* and *Codium isthmocladum* in Jamaica and Florida were nutrient-limited, while *Halimeda opuntia*, *Lobophora variegata*, and *Sargassum polyceratum* did not respond to increased nutrients. Because macroalgae can rapidly assimilate pulses of ammonium (NH₄⁺) and soluble reactive phosphate (SRP) from the water column, macroalgal blooms can occur in oligotrophic tropical and subtropical waters, such as those off Floridian shores (Lapointe et al. 1994; Bell and Elmetri 1995; Duarte 1995).

B.4.3 Role of Nitrogen/Phosphorus Pollution in Macroalgal Growth

While it can be difficult to find a significant correlation between nutrient increase and macroalgal abundance using short-term field studies (McCook 1999), there is limited evidence to provide a linkage to nitrogen/phosphorus pollution loadings from inland waters (i.e., point and nonpoint sources) and the development of macroalgal HABs (Lapointe et al. 2005c). For example, there are studies that use stable N isotope ratios (δ¹⁵N; ‰) in macroalgae that provide evidence that nutrient inputs from inland waters are the primary cause of macroalgal blooms. Lapointe et al. (2004) conducted a study in the Lower Florida Keys using δ¹⁵N signatures of macroalgae and seagrass epiphytes to examine the effects of dissolved inorganic nitrogen (DIN) from local wastewater and agricultural runoff. The results showed that nutrient enrichment from

point and nonpoint sources is one of the main reasons for the seasonal development of macroalgal (as well as phytoplankton, seagrass epiphyte) blooms in the Florida Keys. Another stable N isotope analysis was conducted on macroalgal tissue to examine whether N from secondarily treated wastewater was more influential than N from upwelling in southeast Florida (Lapointe et al. (2005a). The results showed that wastewater sources have contributed more to macroalgal blooms of *Caulerpa branchypus* in southeast Florida in recent years. Lapointe and Bedford (2007) conducted a study in Lee County, Florida, where serious macroalgal blooms of rhodophytes occurred in 2003–2004; using stable N isotope ratios ($\delta^{15}\text{N}$; ‰) in macroalgae to identify sources of N, drift rhodophyte blooms in recent years were determined to result from an increase in point and nonpoint source nitrogen/phosphorus pollution.

Three indirect methods have mainly been used to examine whether N or P or both limit macroalgal productivity: (a) examining the N:P ratios of dissolved inorganic nutrients in the water column; (b) examining N:P ratios of the algal tissue; and (c) conducting N and P enrichment experiments on macroalgae. Each method has limitations, such as not knowing at what point one should sample the water column (nutrient increases can be in pulses or in a steady stream) and the different uptake and storage capacities of nutrients in each species of macroalga (Fong et al. 2003).

It is debated whether N or P is the principal cause of macroalgal blooms (Lapointe et al. 1992). Macroalgae in carbonate-rich waters (such as the Caribbean) have been found to be P-limited, while macroalgae in siliciclastic waters are not (Lapointe et al. 1992). For example, the C:N:P composition of *Codium isthmocladum* in southeast Florida differs markedly from other reefs in the Bahamas, Greater Antilles, and Lesser Antilles. P levels were two-fold lower in the Caribbean region, which means that coastal waters of southeast Florida have reduced P-limitation compared to a stronger P-limitation in Caribbean waters that are rich in carbonate (Lapointe et al. 2005c). Concentrations of NO_3^- and NH_4^+ in near bottom reef waters in Palm Beach and Broward counties were 0.07–8.91 micromolar (μM) and < 0.05 –2.99 μM , respectively, while SRP concentrations were 0.066–0.766 μM (average of > 0.29 μM in southeast Florida) during summer 1996 studies. DIN:SRP ratios averaged 15.3 in the northern area and 14.4 in the southern region, with peak values of > 20 (Lapointe et al. 2005c).

Studies have been conducted to determine direct cause-effect relationships between nitrogen/phosphorus pollution and macroalgae. The following are examples of studies that examined whether N enhancements increased macroalgal productivity:

- Smith et al. (1981) conducted a study that examined macroalgal cover before and after sewage diversion in Kanohe Bay, Hawaii. N increases in the water column increase N uptake rates, N content of the tissue, photosynthesis-irradiance curves and maximum photosynthetic rates, and accelerate the growth of fronds, resulting in increases in macroalgal biomass (Valiela et al. 1997).
- Collado-Vides et al. (2007) used data from a long-term seagrass monitoring program (Fourqurean et al. 2003; Fourqurean et al. 2001) in the Florida Keys National Marine Sanctuary (FKNMS) to examine long-term abundance of macroalgae. The authors suggest that increases in N availability are the primary reason for increases in calcareous green algae

and red algae, which seem to occur more commonly in inshore areas, though they do not disregard other factors (Collado-Vides et al. 2007).

The following are examples of studies that examined whether P enhancements increased macroalgal productivity:

- Lapointe (1989) conducted a study in southern Florida Bay that examined tissue C:N:P ratios and alkaline phosphatase capacity (ability to uptake P). The results showed that tissue C:P and N:P of *Gracilaria tikvahiae* and *Laurencia poitei* (Rhodophyta) ranged from 1,080 to 1,939 and 75 to 147 respectively; the tissue C:P and N:P of phaeophytes examined, *Sargassum polycertium* and *S. pteropleuron* (Phaeophyta) ranged from 550 to 1,307 and 23 to 25, proving P-limitation in such algae.
- Lapointe (1985) conducted a pulsed study in the Florida Keys, finding that P was more important than N as a limiting nutrient in regulating the growth of *Gracilaria tikvahiae* (Rhodophyta).
- Lapointe and Bedford (2010) conducted monitoring studies off the coral reefs of southeast Florida, finding that *Caulerpa brachypus f. parvifolia* (Chlorophyta) blooms were P-limited (N:P of ≈ 37).

Although a handful of studies have been conducted to examine direct impacts of nutrients on macroalgal production, the results vary, as shown below and in Table B-5.

- Lapointe and O'Connell (1989) studied nutrient enrichments (14 days) on the green alga *Cladophora prolifera* (Chlorophyta) in shallow surface waters of Bermuda. Results showed enhancement of productivity, light-saturated photosynthetic capacity, and elevated C:N, C:P and N:P ratios (25, 942, 49 respectively). Although the region was P-limited, an increase in N showed a greater growth rate and photosynthetic capacity compared to P-enrichments. Maximum enhancement of productivity occurred with both N and P additions.
- Lapointe (1997) conducted nutrient enrichment experiments in Jamaica and southeast Florida. The results showed that bottom-up control such as nutrient enrichment “was a major factor causing increased productivity and standing crops of macroalgae on reefs.”
- *Gracilaria edulis* was exposed to nutrient pulses, which resulted in increased tissue N, chlorophyll *a*, and amino acids (Costanzo et al. 2000).
- Miller et al. (1999) conducted nutrient enhancement studies (> 41 days) on an offshore reef in Key Largo, Florida. They did not show that macroalgal or crustose algal abundance had been increased by elevated nutrients above predicted threshold response levels of 1.0 μM DIN and 0.10 μM SRP (Lapointe et al. 1993).
- In an 18-month study conducted at six sites across Florida Bay, Armitage et al. (2005) found that nutrient enhancements had variable responses on macroalgae, but in general, no algal overgrowth occurred.

- Lapointe (1985) conducted an *in situ* cage culture study in the Florida Keys on *Gracilaria tikvahiae* to address N and P dynamics in macroalgae through nutrient pulsation. He found that the concentration (rather than frequency) of nutrient pulses had more pronounced effects on molar ratios of N and P.
- In nutrient enrichment experiments at two sites in Puerto Rico, Fong et al. (2003) found that the previous storage of nutrients reflected how macroalgae responded to enhanced nutrients.
- Fong et al. (1993) conducted a microcosm nutrient enhancement experiment in shallow coastal lagoons on different algal groups (green macrophytes, phytoplankton, and benthic cyanobacterial mats) under five N:P treatments. They found that N directly controlled macroalgal biomass, and when the N supply exceeded macroalgal demand, N was then available to the other algal groups.
- Littler et al. (1986) conducted nutrient enrichment studies in caged experiments in Looe Key National Marine Sanctuary and found that nutrient enhancement did not significantly affect growth rates of macroalgae.
- Schaffelke and Klumpp (1998b) found that increased macroalgal growth was most pronounced by enhancement (24-hour or 1-hour duration) of both N and P for *Sargassum baccularia* in the Great Barrier Reef.

Table B-5. Information on macroalgae relevant to developing potential numeric criteria endpoints

Characteristic	Response	Citation
Threshold concentrations for macroalgal blooms	Belize Barrier reef DIN (1.0µM) and SRP (0.1µM) thresholds noted to sustain macroalgal blooms on Caribbean coral reefs.	(Lapointe et al. 1993; Smith et al. 1981)
Macroalgal biomass	Microcosm experiment to replicate N and P limitation in shallow coastal lagoons. 3 algal groups: Green macrophytes, phytoplankton, and benthic cyanobacteria mats were exposed to 4 levels of nutrient enrichment in 5 N:P ratios. Summer: Biomass stimulated regardless of P supply when N supply rate was between 0.034 – 2.170 mg/L*day; Biomass and nutrient addition had a positive relationship when P was 0.025 – 0.040 mg/L*day, and N was 0.001-0.270 mg/L*day. Spring: P loading rates that stimulated macroalgae was the same as in the summer; stimulation occurred at lower N supply rates.	(Fong et al. 1993)
Transition from eelgrass-dominated to macroalgae-dominated habitats	Hamblin Pond, Massachusetts Transition from eelgrass to macroalgae-dominated habitat occurred from land-derived N loading rate of 30 kg/ha*yr.	(Hauxwell et al. 2001)
Macroalgal growth	Great Barrier Reef, Australia <i>Sargassum baccularia</i> Could become nutrient-sufficient at moderately enhanced nutrient conditions: 3–5 µM ammonium, 0.3–0.5 µM phosphate	(Schaffelke and Klumpp 1998a)

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Characteristic	Response	Citation
	Lab experiments: Promoted growth at 3–5µM ammonium, 0.3–0.5µM phosphate Half maximal growth: 0.5–1µM ammonium, 0.05–0.1µM phosphate (estimated concentrations in culture medium: 0.25–0.5µM ammonium, 0.04–0.07µM phosphate) Estimated maximum growth at: 2–4µM ammonium, 0.25–0.45µM phosphate	
Maximum growth rate of <i>Ulva fasciata</i>	Maximum growth rate of <i>Ulva fasciata</i> occurred with residual NO ₃ ⁻ concentrations of ≈1µM.	(Lapointe and Tenore 1981)
Macroalgal blooms reported for coral reef regions	Hawaii, Kaneohe Bay (before sewage diversion) <i>Dictyosphaeria cavernosa</i> , <i>Acanthophora spicifera</i> , <i>Gracilaria</i> sp. Mean concentrations: DIN 1.13 µM SRP 0.36 µM DIN:SRP= 3:1	(Smith et al. 1981; Lapointe et al. 1997)
Macroalgal blooms reported for coral reef regions	Red Sea, Gulf of Eilat <i>Enteromorpha</i> sp. Mean concentrations: DIN 1.20 µM	(Genin et al. 1995)
Macroalgal blooms reported for coral reef regions	Martinique <i>Sargassum filipendula</i> , <i>Enteromorpha chaetomorphoides</i> Mean concentrations: DIN 1.20 µM SRP 0.25 µM DIN:SRP= 4.8	(Littler et al. 1993)
Macroalgal blooms reported for coral reef regions	Bermuda, Harrington Sound <i>Cladophora prolifera</i> Mean concentrations: DIN 1.66 µM SRP 0.14 µM DIN:SRP= 11.9	(Lapointe and O'Connell 1989)
Macroalgal blooms reported for coral reef regions	Southeast Florida <i>Codium isthmocladium</i> DIN 1.61 µM Mean concentrations: SRP 0.19 µM DIN:SRP= 8.5	(Lapointe 1997)
Macroalgal blooms reported for coral reef regions	Discovery Bay, Jamaica <i>Lobophora variegata</i> , <i>Dictyota</i> spp., <i>Sargassum</i> spp., <i>Chaetomorpha linum</i> , <i>Cladophoropsis</i> sp. Mean concentrations: DIN 12.21 µM (Back-reef station avg: Nov 1987-May1988 7.27µM; April-June 1989 5.67µM) SRP 0.17 µM (Back-reef station avg: Nov 1987-May1988 0.27µM; April-June 1989 0.45µM) DIN:SRP= 71.8	(Lapointe 1997)
Macroalgal blooms reported for coral reef regions	Belize Barrier Reef, Man-O-War Cay (Guano island) <i>Ulva lactuca</i> , <i>C. linum</i> , <i>A. spicifera</i> Mean concentrations: DIN 4.84 µM	(Lapointe et al. 1993)

Characteristic	Response	Citation
	SRP 1.56 μM DIN:SRP= 5.7	
Macroalgal blooms	Southeast Florida <i>Codium isthmocladum</i> Low N:P ratio (<10:1) is critical to sustaining balanced growth during bloom formation	(Lapointe et al. 2005c)
Macroalgal blooms	Southeast Florida <i>Caulerpa brachypus f. parvifolia</i> N:P \approx 37	(Lapointe and Bedford 2010)

The role herbivory plays in the dynamics of macroalgae and coral is a highly debated subject. Miller et al. (1999) conducted a block design field experiment using cages and partial cages to observe the effects of nutrient enrichment on coral reef ecosystems without herbivory. They elevated nutrient levels (1.1–3.1 μM total inorganic N and 0.11–0.14 μM SRP, compared to control samples of 0.3–0.9 μM total inorganic N and < 0.03–0.03 μM SRP) for 41 days without replenishment of nutrients in Key Largo, Florida. They found that the abundance of larger macroalgae had not increased above predicted threshold response levels of 1.0 μM for total inorganic N or 0.10 μM for SRP. The authors also found that, contradicting other studies, N-fixing cyanobacteria were enhanced, not suppressed, by elevated nutrients.

Sotka and Hay (2009) conducted a 20-week, manipulative field experiment in the Florida Keys to assess the “separate and interactive effects of herbivory and nutrient enrichment on the development of macroalgal communities and the fitness of the corals *Porites porites* and *Siderastrea sidereal*.” They found no direct effects of nutrient enrichment, but that total macroalgae and the common seaweeds *Dictyota spp.* were suppressed following nutrient enrichment, but not in the absence of large herbivorous fishes. That observation suggests that fishes were selectively feeding on nutrient-enriched macrophytes. In addition, the study reports that upright macrophytes never exceeded approximately 15 percent cover in the presence of herbivores but increased to 80–100 percent in the absence of herbivores. Nutrient additions did not affect the survivorship of corals, but the researchers found that nutrients stimulated the growth of *P. porites*. Thus, the experiment indicates that the loss of herbivores is a greater factor that drives macroalgal blooms on coral reefs than anthropogenic nutrient inputs. Other studies by Burkepile and Hay (2006; 2009) show that, although nutrients do play a role, macroalgae are more controlled by herbivores.

B.4.4 Scientific Justification for Not Proposing Macroalgae as an Assessment Endpoint

Although there is substantial evidence linking increases in macroalgae to nitrogen/phosphorus pollution (e.g., Collado-Vides et al. 2007; Smith et al. 1981; Lapointe 1985; Lapointe 1989; Lapointe and O'Connell 1989; Lapointe and Bedford 2010), EPA has chosen not to consider a macroalgae endpoint because chlorophyll *a* serves as a better indicator of nitrogen/phosphorus pollution.

B.5 Epiphytes

Epiphytic algal assemblages inhabit the external surfaces of seagrass leaves and shoots. The algae and bacteria in those assemblages are an important component of natural ecosystems, providing energy through grazers to the estuarine food web. Algal assemblage composition has been used for biological assessment of both marine and freshwater systems and in the reconstruction of historical water quality conditions using paleo-reconstruction and alga-specific water quality optima. Because many algal species are highly sensitive to water quality conditions, epiphyte assemblage composition could serve as an indicator of nitrogen/phosphorus pollution impacts that occur before degradation of seagrass density, diversity, or distribution (Frankovich and Fourqurean 1997; Cambridge et al. 2007; Peterson et al. 2007) or other estuarine eutrophication responses. In Florida, where it is difficult to recolonize lost seagrass beds, such an early warning system could be useful for protecting estuarine ecosystem integrity (Bricker et al. 2003a).

B.5.1 Measuring Epiphytes

Epiphytic biomass is typically quantified as chlorophyll *a* using a variety of methods, such as ash-free dry weight or biovolume of algae (e.g., Lapointe et al. 2004; Armitage and Fourqurean 2009). Composition of the assemblage is typically quantified through direct sampling and taxonomic identification to species or lowest practical taxonomic resolution (e.g., Frankovich et al. 2009). Typically, the composition data are analyzed with concurrent or spatially averaged water quality or seagrass data.

B.5.2 Effect of Nitrogen/Phosphorus Pollution on Epiphytes

Epiphytes have been shown to exhibit various patterns in response to nitrogen/phosphorus pollution depending on the pre-enrichment epiphyte community and the baseline ecosystem conditions (e.g., climate, eutrophy/oligotrophy, hydrodynamics). Those responses principally include increased biomass, growth rates, and densities (Armitage and Fourqurean 2009; Armitage et al. 2006; Balata et al. 2008a; Frankovich and Fourqurean 1997; Gil et al. 2006; Herbert and Fourqurean 2008; Jensen and Gibson 1986; Peterson et al. 2007; Raberg and Kautsky 2007; Richardson 2006; Russell et al. 2005; Tomasko and Lapointe 1991) and changes in assemblage composition (Sagasti et al. 2000; Lapointe et al. 2004; Cambridge et al. 2007; Lane and Brown 2007; Peterson et al. 2006; Balata et al. 2008b; Frankovich et al. 2009; Richardson 2006; Richardson et al. 2006). Such direct effects of nitrogen/phosphorus pollution on epiphyte biomass and assemblage structure can, in turn, negatively affect the seagrasses on which the epiphytes grow because of reduced light availability and increased nutrient competition and drag (Sand-Jensen 1977; Orth and Montfrans 1984; Cambridge et al. 1986; Ferdie and Fourqurean 2004; Howard and Short 1986; Twilley 1985; Frankovich and Zieman 2005; Peterson et al. 2007; Lapointe et al. 2004).

A study conducted along a nutrient gradient in Florida Bay (TN ranging from 0.266 to 1.99 mg/L and TP ranging from 0.0065 to 0.052 mg/L TP) discovered that epiphytes were more abundant with higher nutrient concentrations (Frankovich and Fourqurean 1997). In another controlled field study, it was found that phosphorus additions resulted in increased red algae-cyanobacterial complex and green algae and a decrease in diatoms. The same changes were observed but less

pronounced when N and P were added. However, the investigators determined that diatom assemblage from the same site and time, regardless of nutrient treatment, remained more similar to one another than to the diatom assemblage subject to identical nutrient treatments from different sites and times (Frankovich et al. 2009), indicating that temporal and spatial environmental heterogeneity is a stronger determinant of diatom assemblages than nutrients.

Characteristics of the epiphyte assemblage that can be used as endpoints for numeric criteria development include biomass and composition. Epiphyte biomass is known to contribute to seagrass decline, as determined in experimental and survey studies (Cambridge et al. 1986; Howard and Short 1986; Lapointe et al. 1994; Wear et al. 1999). A number of experimental studies have demonstrated increased epiphyte biomass with nutrient enrichment (Tomasko and Lapointe 1991; Lapointe et al. 1994; Valiela et al. 1997; Wear et al. 1999; Balata et al. 2008b), regardless of grazing intensity (Peterson et al. 2007). However, other studies have shown equivocal responses because of a variety of factors, including potential top-down control of epiphyte production by grazers (Armitage et al. 2005; Heck et al. 2006), or weak relationships between nutrients and epiphyte biomass in experiments and surveys (Frankovich and Fourqurean 1997).

Studies relevant to epiphyte endpoint development are listed in Table B-6. Where available, nutrient concentrations associated with responses are noted.

Table B-6. Information on epiphytes relevant to developing numeric criteria endpoints

Characteristic	Response	Citation
Biomass	Sites with elevated NO ₃ -N (0.5-10 µM) and NH ₄ -N (1-3 µM) with increased biomass relative to low NO ₃ -N (< 1.5 µM) and NH ₄ -N (<1 µM)	(Balata et al. 2008b)
Biomass	Increased with N enrichment (from 0.064 to 0.121 µM NO ₃ -N) and especially on an oligotrophic coast in Australia	(Russell et al. 2005)
Biomass	The only explanation for reduced seagrass growth in Australia	(Cambridge et al. 1986)
Biomass	Increase with minor nutrient enrichment regardless of grazer intensity	(Peterson et al. 2007)
Biomass	Increased with experimental N and P addition	(Wear et al. 1999)
Biomass (AFDM percent of blade weight)	Increased in Hyper and Eutr treatments (TN = 19-39 µM; TP = 0.4-1 µM) relative to Oligo treatment (TN=12 µM; TP = 0.2 µM)	(Lapointe et al. 2004)
Biomass and load	Increased with nutrient enrichment additions mimicking waste/stormwater N and P loading rates (1.43 g N m ⁻² day ⁻¹ and 0.18 g P m ⁻² day ⁻¹ , molar N:P ratio 17.6:1)	(Ferdie and Fourqurean 2004)
Chlorophyll <i>a</i>	Increased biomass and shoalgrass decreases	(Heck et al. 2006)

Characteristic	Response	Citation
Composition	Also altered between high and low nutrient sites	(Balata et al. 2008b)
Composition	P addition increases red algae and cyanobacteria taxa and shifts diatom taxa in the summer; N enrichment reduced these responses; nutrient enrichments meant to mimic typical sewage discharge	(Frankovich et al. 2009)
Composition	Increased diatoms and red and brown algae with N and P addition	(Wear et al. 1999)
Growth	Likely a regulator of SAV growth in Sarasota Bay	(FDEP 2008; Tomasko 2008)
Seagrass (<i>Halodule wrightii</i>) growth	Decreased with exclusion of grazers and increase in epiphyte biomass; seagrasses senesced	(Howard and Short 1986)

B.5.3 Scientific Justification for Not Proposing Epiphytes as an Assessment Endpoint

Epiphyte biomass is known to contribute to seagrass decline principally due to shading (Sand-Jensen 1977; Orth and Montfrans 1984; Twilley et al. 1985) and can adversely affect corals (Lapointe et al. 2005b; Lapointe et al. 2005c; Lapointe and Bedford 2010; McCook 1999; Scavia and Bricker 2006). While some studies have demonstrated increased epiphyte biomass with nutrient enrichment, others suggest a variety of causative factors, including potential top-down control of epiphyte production by grazers. Also, factors in addition to nutrients can cause an increase in epiphytes. The use of epiphyte community composition was also explored; however, current data relating epiphyte dynamics to nutrient concentrations are limited both spatially and in number. The development of an enrichment index based on the taxonomic composition of algae could potentially be a worthwhile research effort; however, it is not available at this time.

B.6 Seagrass

Seagrass, a type of submerged aquatic vegetation (SAV), provides habitat for many fish, birds, and invertebrates, and it is an important food source for endangered species such as manatees and green turtles in the State of Florida. Water quality degradation and nutrient enrichment have played a role in the decline of SAV worldwide (USGS 2001). A global assessment discovered that since 1980, seagrasses have been lost at an average rate of 110 km² per year, which is an increase from 0.9 percent per year before 1940 to 7 percent per year since 1990 (Waycott et al. 2009). Like the global study, the trends in Florida's seagrass are similar. Along the Gulf Coast of Florida, six of nine Florida estuaries (summarized in Table B-7) show recent or continued declines in seagrass coverage (USGS 2001), the predominant causes of which are nutrients, dredging, propeller scarring, hydrologic alterations, increased turbidity, and chronic light reduction. During public workshops held by FDEP, presenters reported continued increases in seagrass in Tampa Bay and Sarasota Bay to approximately 12,000 and 5,000 hectares in 2008, respectively, with increases attributed to efforts that are underway in both Tampa Bay and Sarasota Bay to reduce N.

Table B-7. Seagrass trends in coverage area along the Gulf Coast of Florida

Waterbody	Seagrass coverage estimate	Historical seagrass coverage trends	Most common cause of seagrass degradation
Big Bend	250,000 hectares in 1992	51% decrease since 1984	Hydrological alterations, nutrient enrichment
Choctawhatchee Bay	1,722 hectares in 1992	Significant loss between 1949 and 1985, 28% increase since 1983	Eutrophication, propeller scarring from recreational boats
Florida Bay	124,787 hectares in 1994	12% decrease since 1987	Increase turbidity, chronic light reduction
Great Charlotte Harbor	21,802 hectares in 1999	6% decrease from 1982	Increases in turbidity and total suspended solids in the water column, changes to freshwater flows, dredging
Pensacola Bay	1,814 hectares in 1992	52% decline since 1960	Sewage/industrial waste, dredging activities, beachfront alteration
Peridido Bay	120 hectares in 2003	74% decline since 1940	Nutrient loading, wastewater effluent, marina, docks, GIWW dredging
Sarasota Bay	3,715 hectares in 2002 ^a	5% increase since 1988	Rapid urbanization, wastewater treatment
St. Andrew Bay	3,979 hectares in 1992	15% decrease since 1964	Atmospheric deposition of nutrients, stormwater runoff, propeller scarring
Tampa Bay and St. Joseph Sound	10,554 hectares in 2002 ^a	35% decrease from 1950 to 1982. 16% increase since 1982	Rapid population growth, dredging activities, nonpoint pollution

Source: Adapted from USGS (2001)

^a Presentations at the St. Petersburg workshop (February 4, 2010) reported ~12,000 and ~5,000 hectares of seagrass for Tampa Bay and Sarasota Bay in 2008, respectively.

B.6.1 Effect of Nitrogen/Phosphorus Pollution on Seagrasses

The literature shows that nitrogen/phosphorus pollution play a substantial role in the decline of seagrass, but the effect is indirect via their effect on the availability of light and dissolved oxygen (see Table B-1 and Table B-2). First, nutrients increase phytoplanktonic turbidity, which absorbs light, thus limiting seagrass exposure to it, leading to impaired photosynthetic processes, reduced

growth, and increased mortality. This effect can be exacerbated by suspension of abiotic substrate exposed to turbulence because of seagrass loss, causing further increases in turbidity and decreases in light availability (a positive feedback loop). Second, excess epiphytic algal growth in response to nutrients has also been implicated in shading seagrasses and affecting seagrass structure. Also, decay of phytoplankton blooms caused by nutrient enrichment consumes oxygen that would have been available for SAVs. A Chesapeake Bay study determined that healthy seagrasses grow when chlorophyll *a* concentrations are less than 15 µg/L (Boynton et al. 1996).

B.6.2 Factors that Affect Seagrass Growth

Light availability is a measure of how much light reaches the SAV. It is largely determined by color and water clarity, which is affected by chlorophyll *a* and macroalgal abundance. When sunlight reaches the water surface, some of that light is reflected, while the rest penetrates the water. Particulate matter in the water column, including chlorophyll *a*, attenuates another portion of the light, resulting in an even smaller fraction that reaches the plants on the bottom. For example, Tomasko et al. (2001) estimated that phytoplankton biomass contributes an average of 29 percent light attenuation within the Lemon Bay water column and is largely a factor of chlorophyll *a* concentrations. One study found that certain species of seagrass do best when algal mats are smaller than 0.25 m in diameter (Holmquist 1997), because light can penetrate to the SAVs. If any epiphytes are growing on the SAV, the light availability is further restricted (McLaughlin and Sutula 2007).

Most seagrasses have a minimum light requirement of 10 percent surface irradiance (Duarte 1991). Other studies conducted on seagrasses in the Indian River Lagoon System in Florida found that the minimum annual light requirement was 20 percent surface irradiance (Steward et al. 2005) while another found a minimum of 24 percent to 37 percent irradiance for two species of seagrasses (Kenworthy and Fonseca 1996); another study conducted in Tampa Bay showed a light requirement of 20.5 percent (Greening and Janicki 2006).

B.6.3 Habitat Restrictions

Some studies provide summaries of SAV habitat restrictions (Table B-8) and empirical relationships between SAV depth limits and abundance as a function of various eutrophication factors (Tables B-9 and B-10) (Krause-Jensen et al. 2008; McLaughlin and Sutula 2007). Those summary tables include regression relationships developed for Sarasota Bay, Charlotte Harbor, and Tampa Bay (Greening and Janicki 2006; Tomasko et al. 1996; Tomasko and Hall 1999).

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Table B-8. Habitat restrictions of submerged aquatic vegetation (SAV)

SAV species	Factor leading to decline in SAV habitat	Thresholds	Source(s)
Chesapeake Bay Seagrasses <i>Zostera marina</i> and <i>Ruppia maritima</i>	Light Extinction Coefficient	< 1.5 /meter	Boynton et al. (1996)
Chesapeake Bay Seagrasses <i>Zostera marina</i> and <i>Ruppia maritima</i>	Chlorophyll <i>a</i>	Concentration of < 15 µg/L	Boynton et al. (1996)
Chesapeake Bay Seagrasses <i>Zostera marina</i> and <i>Ruppia maritima</i>	TN	Concentration of < 10 µM	(Boynton et al. 1996)
<i>Halodule wrightii</i> <i>Zostera marina</i>	Light Limitation	> 8%–18% surface light > 10%–20% surface light > 5–6 hours of light saturated irradiance	(Duarte 1991; Dunton 1996; Zimmerman 1994; Zimmerman et al. 1995b)
<i>Halodule wrightii</i> and <i>Syringodium filiforme</i>	Light Limitation	Between 24 and 37% iridescent	(Kenworthy and Fonseca 1996)
<i>Halodule wrightii</i> <i>Zostera marina</i>	Light Limitation	Minimum growing season light requirement: 20% surface light	(Steward et al. 2005)
<i>P. pectinatus</i>	Sulfide Toxicity	0.48 - 1.27 mg/g sediment sulfide	(van Wijck et al. 1992)
SAV habitat	Sediment Grain Size	<20% silt and clay	(Koch 2001)
SAV habitat	Sediment Organic Carbon	<5 %	(Koch 2001)
Seagrass meadow: <i>Thalassia testudinum</i> (turtle grass), <i>Halodule wrightii</i> (shoal grass), and <i>Syringodium filiforme</i> (manatee grass)	Macroalgal Biomass Cover	Macroalgae patches >0.25 m diameter	(Holmquist 1997)
<i>Vallisneria americana</i>	Light Limitation	Low Salinity: 9% surface light High Salinity: 14% surface light	(Dobberfuhl 2007)
<i>Zostera marina</i>	Macroalgal Canopy Height	Canopy height > 9-12 cm	(Hauxwell et al. 2007)
<i>Zostera marina</i>	Ammonia Toxicity	25 µm in 5 weeks 125 µm in 2 weeks	(van Katwijk et al. 1997)
<i>Zostera marina</i>	Sulfide Toxicity/ Dissolved (McLaughlin and Sutula 2007)Oxygen Concentrations	water column DO < 7.3 kPa (30 - 35% saturation)	(Pedersen et al. 2004)
<i>Zostera marina</i>	Sulfide Toxicity	< 0.4 mM sediment sulfide	(Goodman et al. 1995)
<i>Zostera marina</i>	Sulfide Toxicity	<2 mM sediment sulfide	(Pregall et al. 1984; Smith et al. 1988)
<i>Zostera noltii</i>	Ammonia Toxicity	16 µm in 16 days instantaneous 200 µm	(Brun et al. 2002)

Source: adapted from McLaughlin and Sutula (2007)

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Table B-9. Literature compilation of empirical relationship between depth limits of seagrass and predictor variable related to eutrophication

Dependent variable	Independent variables	Regression equation	Summary data	Location (depth, D=deep, S=shallow)
Depth limit, Z_c (m) of seagrasses				
<i>Halodule wrightii</i> , <i>Syringodium filiforme</i>	Attenuation coeff., K (m^{-1})	Not defined. $I_{Z_c} = I_0 e^{-kZ_c}$ range: 24–37%, highest in turbid waters	R^* , $N=6$	Indian River Lagoon, FL, USA (D) (Kenworthy and Fonseca 1996)
<i>Thalassia testudinum</i>	Secchi depth, S (m^{-1})	$Z_c = 0.581 + 0.789S$	$R = 0.67^{**}$ $N=11$	Puerto Rico, USA (D) (Vicente and Rivera 1982)
<i>T. testudinum</i>	Chl a ($C_{t,s}$) avg. for month t and Bay segments	$\ln C_{t,s} = \alpha_{t,s} + \beta_{t,s} \ln(Z_{t,s})^a$	$R^2 = 0.67$	Tampa Bay, FL, USA (D) (Greening and Janicki 2006)
Various seagrasses	Attenuation coeff., K (m^{-1})	$\log Z_c = 0.26 - 1.07 \log K$ $\rightarrow Z_c = 1.86/K$	$R^2 = 0.77^{***}$ $N=72$	Across continents (D) (Duarte 1991)
<i>Zostera capricorni</i>	Secchi depth, S ; light, I ; total nitrogen, TN; Susp. solids, S_s ; Chlorophyll, Chl	Not defined. Variables analyzed individually	S : $R^2 = 0.78^*$ I : $R^2 = 0.87^*$ TN: $R^2 = 0.91^*$ S_s : $R^2 = 0.93^*$ Chl: $R^2 = 0.88^*$ $N=8$	S. Moreton Bay, Queensland, Australia (D) (Abal and Dennison 1996)
<i>Z. marina</i>	Attenuation coeff., K (m^{-1})	$Z_c = 1.62/K$	No info.	Woods Hole, MA, USA (D) (Dennison 1987)
<i>Z. marina</i>	Attenuation coeff., K (m^{-1})	$\log Z_c = 0.27 - 0.84 \log K$	$R^2 = 0.40^{***}$ $N=29$	Across continents (D) (Duarte 1991)
<i>Z. marina</i>	Secchi depth, S (m^{-1})	$Z_c = 1.259 + 0.425S$	$R^2 = 0.413^*$ $N=10$	Denmark (D) (Olesen 1996)
<i>Z. marina</i>	Total nitrogen, TN ($\mu g/l$) Secchi depth, S (m^{-1})	$\ln Z_c = 6.039 - 0.755 \ln TN$ $Z_c = 0.339 + 0.786S$	$R^2 = 0.55^{***}$ $N=128$ $R^2 = 0.61^*$ $N=101$	Denmark (D) (Nielsen et al. 2002a)
<i>Z. marina</i>	Water depth, Z_{max} (m^{-1}), Secchi depth, S (m), ln winter [NH_4^+], NH_{4W}^+ (μM)	$Z_c = 2.44 + 0.11Z_{max} - 0.87 NH_{4W}^+ + 0.26S_w$ $Z_c = 1.27 + 0.06Z_{max} - 0.15 NH_{4W}^+ + 0.38S_s$	$R^2 = 0.71^{***}$ $N=46$ $R^2 = 0.42^{***}$ $N=390$	Denmark 1994 (D) (Greve and Krause-Jensen 2005) Denmark 1989–1998 (D)
<i>Z. marina</i>	Secchi depth, S (m-l) Total phosphorus, TP (μM) Total nitrogen, TN (μM)	$Z_c = -0.97 + 1.06S$ $Z_c = -5.361 TP^{-1.86}$ $Z_c = 659.4 TN^{-1.456}$	$R^2 = 0.668^{***}$ $R^2 = 0.602^{***}$ $R^2 = 0.632^{***}$	Denmark 1989–2001 (D) (Sagert et al. 2005)
<i>Z. marina</i>	Attenuation coeff., K (m^{-1})	$K < 0.27 m^{-1}$; $\log Z_c = 0.34 - 1.60K$ $K > 0.27 m^{-1}$; $\log Z_c = 0.10 - 1.02K$	$R^2 = 0.79^{***}$ $R^2 = 0.63^{***}$	Across continents (D) (Duarte et al. 2007)

Source: (Krause-Jensen et al. 2008)

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Table B-10. Literature compilation of empirical relationships between abundance of seagrass and predictor variables related to eutrophication

Dependent variable	Independent variable	Regression equation	Summary data	Location (depth, D=deep, S=shallow)
Seagrass biomass 'attenuation', K _{bio} (m ⁻¹)	Attenuation coeff., K (m ⁻¹)	$\log K_{bio} = 0.19 + 1.4 \log K$	R ² = 0.91*** N= 9	Across continents (D) (Duarte 1991)
SAV-presence/absence	DIN, DIP, Susp. solids, Chl a	No regression data	No info.	Chesapeake Bay, USA (S) (Dennison et al. 1993; Stevenson et al. 1993)
Seagrass areal blade biomass, ABB (g dw m ⁻²)	N-load (kg N day ⁻¹)	$ABB = 336.15 (N\text{-load})^{0.342}$	R ² = 0.536***	Sarasota Bay, FL, USA (S) (Tomasko et al. 1996)
Seagrass areal blade prod. ABP (g dw m ⁻² day ⁻¹)	N-load (kg N day ⁻¹)	$ABP = 11.35 (N\text{-load})^{0.371}$	R ² = 0.536***	Sarasota Bay, FL, USA (S) (Tomasko et al. 1996)
Seagrass community leaf biomass	Sediment silt and clay (%)	Non-linear threshold relation	R = - 0.35*	Phillippines, Thailand (D, <3m) (Terrados et al. 1998)
<i>Zostera marina</i> biomass B (g m ⁻²)	N-load	Negative trend, no regression data	R ² = 0.59***	Waquoit Bay, USA (S) (Lyons et al. 1995)
<i>Z. marina</i> biomass B (g DW m ⁻²)	N-load (kg N ha ⁻¹ year ⁻¹)	$B = 94(10^{-0.029N\text{-load}}) - 1$	R ² = 0.82* N= 7	Waquoit Bay, USA (S)(Hauxwell et al. 2003)
<i>Z. marina</i> density D, shoots (m ⁻²)	N-load (kg N ha ⁻¹ year ⁻¹)	$D = 647(10^{-0.040N\text{-load}}) - 1$	R ² = 0.88* N= 7	Waquoit Bay, USA (S) (Hauxwell et al. 2003)
<i>Z. noltii</i> biomass, BZ (g AFDW m ⁻²)	Photoperiod, Ph (min); residence time, WRT (h); NH ₄ ⁺ , (mg L ⁻¹); org. matter, OM (%); Enteromorpha or Ulva biomass, B _E or B _U (g AFDW m ⁻²)	$\log B = - 1.677 + 2.493(\log Ph) - 3.16 (\log WRT) - 3.902(\log NH_4^+) + 1.593(\log OM) - 0.948(\log B_E) + 0.552(\log B_U)$	R ² = 0.531*** d.f. = 139	Mondego estuary, Portugal (S) (Silva-Santos et al. 2006)
<i>Thalassia testudinum</i> prod., P (g dw m ⁻² day ⁻¹)	Salinity, Sal (per mille); temperature, T (8 °C); subsurface irradiance, I (%)	$P = - 2.151 + 0.045 Sal + 0.070TSal$ and I: positively correlated	R ² = 0.42** d.f. = 55	Charlotte Harbor, FL, USA (S) (Tomasko and Hall 1999)
Dense vegetation (D)	Photic depth Silt Clay Sand Chl a	Positive trend, no regression data Negative trend, no regression data Negative trend, no regression data Positive trend, no regression data Negative trend, no regression data	R ² = 0.16* R ² = 0.23** R ² = 0.34* R ² = 0.23* R ² = 0.25*	Gulf of Mexico (S) (Livingston et al. 1998)
Log odds of observing	Light, PFD (% surface PFD);	$L_j(x) = a_j - 0.047 PFD + 0.062 \times 1_{(PFD > 60)}$	R ² = 0.20* N= 288	Denmark (S, 0–1 m)
<i>Z. marina</i> at j(0%, ≤2%, ≤25%, ≤50%, ≤75%) cover (L _j (x))	exposure, REId; salinity, S	$x (PFD - 60) + 1.218 REId - 0.055S$ $L_j(x) = a_j - 0.081 PFD + 0.108 \times 1_{(PFD > 40)} \times (PFD - 40) + 1.485 REId$ $L_j(x) = a_j - 0.067 PFD + 1.727 REId - 0.085S$ $L_j(x) = a_j - 0.126 PFD + 0.198S - 1.168 \times 1_{(depth < 6)}$	R ² = 0.14* N= 148 R ² = 0.20* N= 138 R ² = 0.38* N= 161	(Krause-Jensen et al. 2003) Denmark (S, 1–2 m) Denmark (D, 2–4 m) Denmark (D, 4–8 m)
SAV vs. plankton	Nutrient load	No relationship found	Non-significant	30 estuaries, across continents (S) (Nixon et al. 2001)
SAV vs. plankton	Nutrient load, conc., residence time, Chl, bathymetry	A correlation matrix	No info.	10 shallow bays, Baltic Proper (S) (Dahlgren and Kautsky 2004)

Source: Krause-Jensen et al. (2008)

One conclusion of those papers is the value of local calibration of empirical models to address site-specific issues such as water color, measured as colored dissolved organic matter (CDOM). Using data collected from Tampa Bay and Charlotte Harbor, investigators developed an empirical model that can be used to estimate an adjusted attenuation coefficient as a function of color, turbidity, and chlorophyll *a*. More recent studies have used this model to develop nomographs for proposing water quality targets for Charlotte Harbor (Corbett and Hale 2006). An application of that concept is shown in Figure B-1, which demonstrates turbidity and chlorophyll *a* levels to achieve selected seagrass depth maxima based on PAR, which is used to evaluate the effect of light on plant growth and color. To achieve PAR levels of 0.25 (and assuming an average maximum seagrass depth of 1.5 m and color of 0 platinum-cobalt units [PCU]), chlorophyll *a* and turbidity results would need to fall below the solid black line shown in Figure B-1. Kelble et al. (2005) investigated PAR attenuation in Florida Bay as a function of CDOM, tripton, and phytoplankton. Several other researchers note that some assemblage characteristics, such as composition, shoot density, and elemental composition, might also be useful endpoints (Bostrom et al. 2002; Cabaco et al. 2009; Howard and Short 1986; Tomasko et al. 1996).

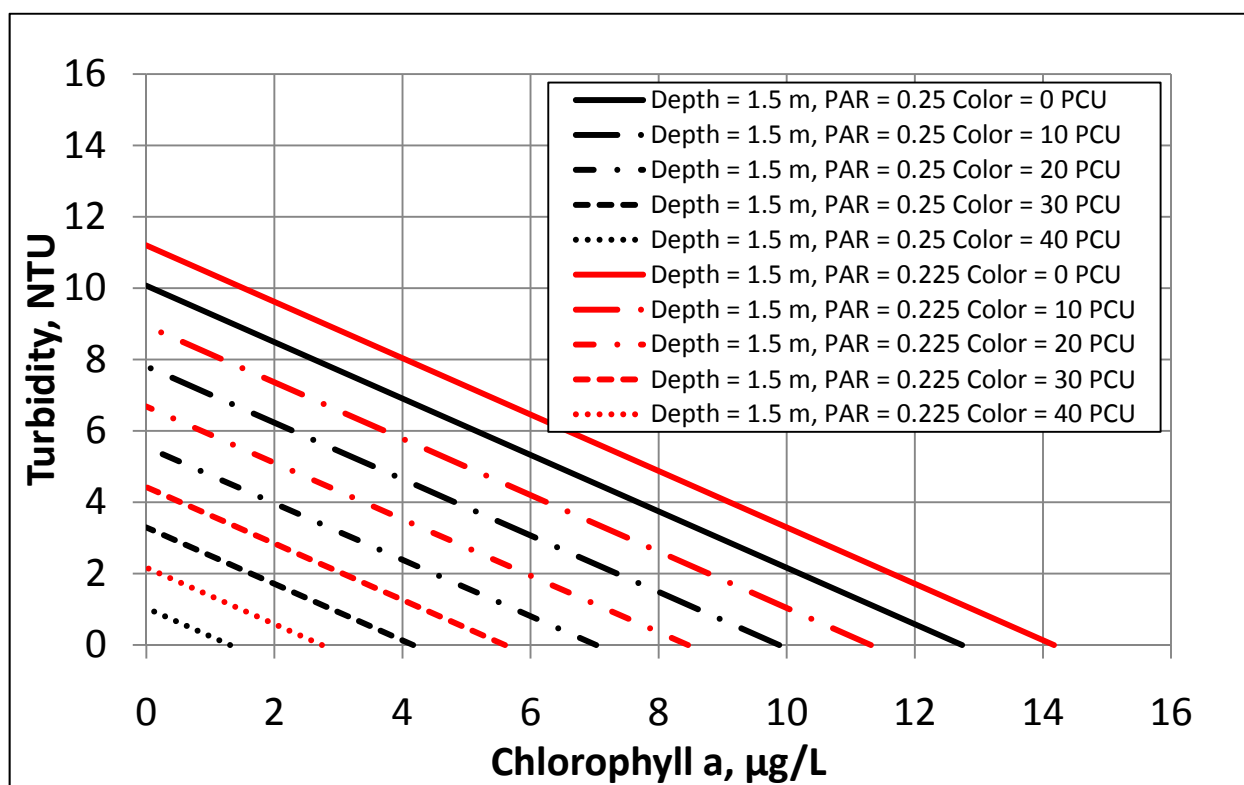


Figure B-1. Nomograph depicting turbidity and chlorophyll *a* levels needed to achieve selected seagrass maximum depths, PAR, and color using empirical model developed from Tampa Bay and Charlotte Harbor data (Corbett and Hale 2006; McPherson and Miller 1994). To achieve a PAR of 0.25 (and assuming an average maximum seagrass depth of 1.5 m and color of 0 PCU), chlorophyll *a* and turbidity results would need to fall below the solid black line.

Regression analyses have also been used to directly estimate watershed loadings for the Indian River and Banana River lagoons (Steward et al. 2005). Watershed loadings of approximately

2.4–3.2 kg/ha/yr TN, 0.41–0.64 kg/ha/yr TP, and 48–64 kg/ha/yr TSS were estimated to meet desired depth limits for seagrass.

B.6.4 Potential Assessment Endpoints

EPA is considering establishing seagrass depth targets for which a selected percent of surface irradiance must be achieved (see Chapter 2).

B.7 Benthic Macroinvertebrate and Fish Indices

Fish and benthic macroinvertebrates include many commercially and recreationally important species, as well as their food resources. Sport fishing, commercial fishing, recreational fishing and shell fishing are important contributors to Florida's economy. Benthic macroinvertebrates are organisms lacking a backbone, large enough to be retained with a 595 micron mesh, and that make their living primarily on the floor bottom, such as sponges, mollusks, segmented worms, crabs, and shrimp.

Indices that measure the biological condition of a waterbody have been developed for benthic macroinvertebrates in Gulf Coast estuaries (e.g., Engle et al. 1994; Macauley et al. 2002; Engle and Summers 1999a). The indices were calibrated to respond to sediment contamination, sediment toxicity, and hypoxia, but they often cannot contribute to isolating or diagnosing a single cause from among these stressors because of confounding factors. Although similar indices for estuarine fish have been developed elsewhere (e.g., Deegan et al. 1997; Harrison and Whitfield 2004; Meng et al. 2002), EPA has not found a fish community condition index that would be applicable to Florida estuaries to isolate the effect of nutrients.

B.7.1 Effects of Nitrogen/Phosphorus Pollution on Benthic Macroinvertebrates and Fish

Fish and benthic macroinvertebrate communities show two fundamental responses to nutrient enrichment and eutrophication. This first is increased secondary production (increased biomass, possibly faster growth) as the community responds to increased algal food base of the food chain, as long as there is no hypoxia. Such responses are summarized in Table B-11. The second is the response to hypoxia: after hypoxia develops, fish avoid the hypoxic waters (usually bottom waters, but sometimes extending through the entire water column), and motile benthic macroinvertebrates also avoid or escape from the hypoxic zone. Those responses are discussed in the sections above (see Table B-2). The remaining macroinvertebrate assemblage changes in response to hypoxia: species tolerant to hypoxia become dominant, total benthic biomass is often reduced, and groups such as deep-burrowing species might disappear. Burrowers are important ecological species because they process sediment and promote aeration in the top layers of sediment. Loss of bioturbation by burrowers and hypoxic bottom waters, in turn, promote severe reducing conditions in the sediment, which has further consequences on sediment biogeochemistry (release of toxic hydrogen sulfide, release of soluble reactive P (SRP), and release of ammonia (e.g., Kemp et al. 2005; McCarthy et al. 2008; Middelburg and Levin 2009; Diaz and Rosenberg 2008). Those biological effects and consequences of hypoxia are discussed in the DO section, above.

In coastal waters, the presence of a hypoxic dead zone results in a lost of habitat for most aquatic organisms. Benthic communities are greatly altered inside the hypoxic zone, and bottom-feeding fish such as flounder are excluded (Switzer et al. 2009). While there may be few species within the hypoxic zone (which causes these areas to be named “dead zones”), on the margins of the hypoxic zone, both fish and benthic macroinvertebrates can be found at higher densities and have higher biomass production (see Table B-9). Pelagic (open-water) fish assemblages are often not affected since they often stay above the hypoxic zone, but their habitat is reduce in terms of their depth range because they avoid swimming to depths at which hypoxic conditions occur (Switzer et al. 2006; Hazen et al. 2009). See Table B-10.

In estuaries, the combined effects of SAV loss and increased phytoplankton production result in a shift in the system food web and energy base toward greater dominance of phytoplankton production, planktonic filter-feeding, and microbial decomposition. That shortens the food chain and results in less energy flowing toward large benthic organisms such as oysters and to large predators (cod, flounder, striped bass) (e.g., Diaz and Rosenberg 2008).

Table B-11. Summary of selected literature on responses and role of fish and benthic invertebrates to estuarine and coastal eutrophication.

Response	Cause/ predictor	Location Estuary (E) or Coastal (C)	Equation Describing Relationship?	Synopsis	Reference
Algae (epiphytes)	fish grazing	Indian River Lagoon, FL (E)	No	Fish grazers can limit some epiphyte growth on seagrass	(Gacia et al. 1999)
Algae (epiphytes)	nutrients, grazers (benthos)	Florida Bay (E)	No	Grazers might be able to control periphyton under moderate enrichment	(Gil et al. 2006)
Algae, benthos (snails)	nutrients	Tidal flats in Mugu Lagoon, southern California (E)	No	Nutrient enrichment promoted cyanobacteria; Increased mortality of snails feeding on cyanobacteria	(Armitage and Fong 2004)
Benthos	pollution gradients	Narragansett Bay (E)	No	More eutrophic areas have simplified benthic community with opportunistic, early colonizers, small body size, no deep burrowers.	(Calabretta and Oviatt 2008)
Benthos	nutrients, hypoxia, salinity, depth, sediment	Northern Baltic Sea (E, C)	No	Uninterpretable; too many collinear variables within the eutrophied Baltic Sea	(Kotta et al. 2007)
Benthos (shrimp)	salinity, temp, rain fall, water level, and wind speed	Florida Bay (E)	No	Florida Bay is critical nursery for pink shrimp fishery	(Browder et al. 1999)
Benthos, fish, zooplankton	nutrients	Chesapeake Bay (E)	No	Comprehensive review of Chesapeake Bay eutrophication	(Kemp et al. 2005)
Coral, benthos (mollusc)	nutrients	Florida Reef Tract (C)	No	Decline of corals over 37 yr period	(Lidz and Hallock

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Response	Cause/ predictor	Location Estuary (E) or Coastal (C)	Equation Describing Relationship?	Synopsis	Reference
					2000)
Fish	nutrients	Everglades National Park (E)	No	Not interpretable	(Green et al. 2006)
Fish	nutrients	Ria Formosa, Portugal (E)	Unknown	Reduction in nut loading caused decline of mullets	(Ribeiro et al. 2008)
Fish	seagrass, algae	Tampa Bay, Florida (E)	No	Small fish prefer habitats with cover (seagrass or drift algae) over open water	(Ryadene and R.E. Matheson 2003)
Fish (growth)	nutrient loading	Chesapeake Bay (E)	Model	Nutrients promote phytoplankton production, which increases menhaden growth in spring, but not in summer	(Brandt and Mason 2003)
Fish (Density, Richness)	Seagrass	FKNMS (C)	No	Structural complexity of seagrass habitat enhances fish diversity	(Acosta et al. 2007)
Fish, benthos	seagrass, temperature	Southern Australia (E)	Yes	Secondary production of crustaceans and fish higher in seagrass habitat than outside; species diversity of fish higher in seagrass habitat.	(Edgar and Shaw 1995)
Fish, benthos	nutrients	Baltic Sea, North Sea (C)	No	nut enrichment increase secondary prod, unless hypoxia appears	(Nixon and Buckley 2002)
Fish, benthos	Phyto- plankton (shading)	West Florida shelf (C)	Yes (model)	Model of trophic cascade.	(Okey et al. 2004)
Phytoplankton	benthos (oysters)	Chesapeake Bay (E)	No	potential effects of native oyster restoration on DO, chlorophyll, light attenuation, and submerged aquatic vegetation. Requires 10x increase of existing oyster biomass	(Cerco and Noel 2007)
Phytoplankton	benthos (suspension feeding)	Chesapeake Bay (E)	Yes (model)	removal of algal biomass and nutrients by filterers is limited by estuarine circulation and depth	(Gerritsen et al. 1994)
Phytoplankton	(benthos) sponges	Florida Bay (E)	No	Sponge die-off in Florida Bay is sufficient to explain phytoplankton standing crop increase without nutrient enrichment	(Peterson et al. 2006)
Phytoplankton, fish, bacteria	nutrients	Long Island Sound (E)	No	Nutrient enrichment increases phytoplankton, microbial and also crust zooplankton, but fish density reduced. Fish not	(Capriulo et al. 2002)

Methods and Approaches for Deriving Numeric Criteria for Nitrogen/Phosphorus Pollution in Florida's
Estuaries, Coastal Waters, and Southern Inland Flowing Waters

Response	Cause/ predictor	Location Estuary (E) or Coastal (C)	Equation Describing Relationship?	Synopsis	Reference
				food limited, might be limited by tox, habitat loss. Excess zooplankton production stimulated microbial loop and hypoxia.	
Primary, secondary producers	nutrient and trace element loadings	Mesocosm experiments (E)	No	Nutrients increase secondary production	(Bundy et al. 2003)
Productivity, overall trophic response	changes in flow, drought	Apalachicola Bay system - Gulf of Mexico estuary (E)	Yes	Apalachicola Bay productivity determined by river flow; 2-3 yr time lags	(Livingston et al. 1997)
Seagrass, algae, benthos, fish	Nutrients	Narragansett Bay (E)	No	Review of Narragansett Bay responses over time. Benthic community changes due to eutrophication and hypoxia.	(Deacutis 2007)
Seagrass, algae, benthos, fish	nutrients	Plum Island Sound estuary in New England (E)	No	Experimental application of nutrients to marsh: responses nonlinear and not always predictable; marsh is susceptible; increase in productivity in low marsh, but decrease of benthos under predator removal.	(Deegan et al. 2007)
Seagrass, benthos	N and P	Waquoit Bay Watershed, Cape Cod, Massachusetts (E)	No	Causal analysis identified nutrients as major stressor	(Serveiss et al. 2004)
Seagrass, fish, benthos	nutrients, salinity	Coastal Baltic Sea (E)	No	Uninterpretable; small patch of Zostera hanging on in Baltic for unknown reasons	(Bostrom et al. 2002)
Sediment composition	nutrients	Reunion Island (C)	No	Nutrient increase leads to coralline algae replacing corals	(Chazottes et al. 2008)

B.7.2 Scientific Justification for Not Proposing Macroinvertebrate and Fish Indices as Assessment Endpoints for Estuarine and Marine Systems

Indices that measure the biological condition of a water body have been developed for benthic macroinvertebrates in Gulf Coast estuaries and near-coastal waters (Engle and Summers 1999; Engle et al. 1994); and they are currently being re-calibrated using a larger dataset developed by the Gulf of Mexico Alliance (GOMA). The existing index was calibrated to detect response to sediment characteristics, contamination, sediment toxicity, and hypoxia and, thus, may not be useful for segregating single causes among these stressors. However, it should be noted that benthic indices of biological integrity (IBI) have been developed for other estuaries, occasionally detecting changes that are most likely due to elevated nutrients (for research in Chesapeake Bay,

see Weisberg et al. 1997; Dauer et al. 2000). Indices for estuarine fish have been developed elsewhere, but EPA has not identified a fish community condition index that would be applicable to Florida estuaries.

B.8 Harmful Algal Blooms

Of the 30 toxic microalgal species in the Gulf of Mexico, one species, *Karenia brevis*, produces frequent blooms that have significant implications for human health and aquatic life (Steidinger et al. 1998).¹ While most blooms occur along the Gulf Coast of Florida, *K. brevis* blooms can also affect the State's Atlantic Coast. This organism produces brevetoxin, a potent neurotoxin that can cause respiratory irritation (at 10^3 cells/L) through aerosolization, neurotoxic shellfish poisoning through consumption of contaminated shellfish (at $> 5 \times 10^3$ cells/L), and large fish kills (at 1 to 2.5×10^5 cells/L). Mortalities of shore birds and manatees have also been linked to *K. brevis* blooms (Landsberg 2002). Occurrence of *K. brevis* from 2002 to 2007 is depicted in Figure B-3.

Although much attention has been given to *Karenia* spp., several other toxic microalgal species are of note. *Gambierdiscus toxicus* is found from Palm Beach to the Dry Tortugas and Florida Bay. *G. toxicus* is suspected to be the cause of fish kills and disease events, and it produces a toxin called ciguatoxin, which causes ciguatera fish poisoning. In laboratory studies, some strains of *G. toxicus* when associated with bacteria are able to take advantage of organic and inorganic N sources on shorter time scales to support future growth.

Lyngbya majuscula blooms are common in Florida, with noted blooms in 1999, 2000, and 2002 across Charlotte Harbor, Cedar Key, Sebastian Inlet, Sarasota Bay, Tampa Bay, Terra Ceia Bay, Palma Sola, Manatee River, and northwest Bradenton. *L. majuscula* can form large, floating mats. To date, no adverse effects on seagrass, fish, or other marine organisms have been reported from *L. majuscula* blooms in Tampa Bay (FWRI N.d.-a). In one study, Paerl et al. (2008) found that when *K. brevis* and *L. majuscula* co-occur, as in the July 2006 bloom near Sanibel and Captiva Islands, that N additions using in situ bioassay promoted *K. brevis* growth while N+P additions promoted *L. majuscula* most strongly. The authors state that the result confirms previous studies that when non-N-fixing HABs and N-fixing cyanobacterial HABs co-occur, both N and P inputs need to be considered. In waters sampled on nearshore reefs in Broward County, investigators identified three distinct *Lyngbya* spp. (Sharp et al. 2009).

There are other HAB species in Florida, as well. *Aureoumbra lagunensis* (Texas brown tide) has been found in low abundance (< 200 cells/mL) at stations in Florida Bay from November 1996 to June 1997 (Muhlstein and Villareal 2007). *Pfiesteria piscicida* and *P. shumwayae* are widely distributed, and some strains have been shown to be toxic to fish (Steidinger N.d.).

¹ While *K. brevis* is the most well-known and studied species, researchers have identified four other *Karenia* species (Haywood et al. 2004) that appear to contribute to red tide blooms (Anderson et al. 2008).

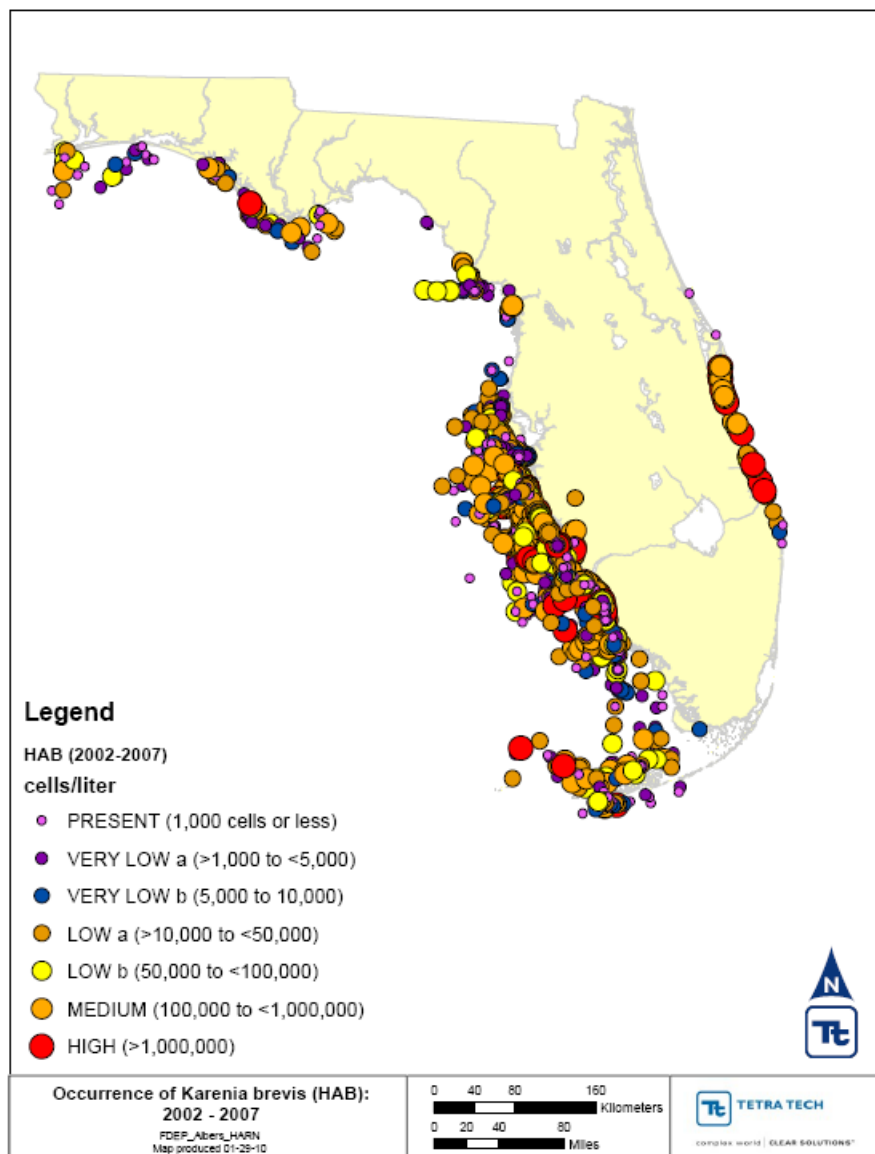


Figure B-3. Occurrence of *K. brevis* from 2002 to 2007 using historic HAB event data from the Fish and Wildlife Research Institute's Marine Resources GIS Internet Map Server

Karenia spp. have been part of the Gulf of Mexico's ecosystem for at least 125 years and probably longer (Steidinger 2009). The earliest reports by European explorers in 1530 described discolored waters and the presence of phycotoxins (Magana et al. 2003). They further reported that native people chronicled their activities by the occurrence of fish kills and red waters (Steidinger et al. 1998). More accounts of blooms and their effects were found between 1844 and 1936. By the mid 1900s significant problems with fish kills, aerosolized toxins, and discolored waters (up to 5.6×10^7 cells/L over 11 months) prompted studies that finally led to the identification of the causative organism. Yet because of inconsistencies in sampling locations and effort, it is unclear whether there is a trend in the frequency of *K. brevis* blooms (FWRI N.d.-b).

Of the HAB species known to exist in Florida, a great deal of laboratory and field research over several decades has clearly shown that *K. brevis*, an athecate dinoflagellate, is highly adapted to conditions in the Gulf. The organism is particularly suited to coastal salinities (33–36 ‰), light conditions and the temperature regime of the West Florida Shelf region (Tester and Steidinger 1997). Before 1973, blooms were thought to initiate inshore around passes because their effects are visible in those areas (Steidinger et al. 1998). The present understanding of bloom dynamics, however, indicates that initiation occurs in the West Florida Shelf about 18–74 km (11–46 mi.) offshore in 12- to 37-m water depth between Tarpon Springs/Clearwater and Ft. Myers/Naples. About 70 percent of outbreaks take place in late summer to fall associated with meanders of the Loop Current in the Gulf. Approximately 17 percent of blooms occur from winter to spring. Following 2–8 weeks of growth, the bloom moves across the shelf toward the coast where the distribution is influenced by winds and along-shore currents. The bloom can become patchy and concentrated, but they can also spread and cause effects over a large portion of the Gulf Coast. For example, the 2005 bloom at its maximum stretched from the Panhandle to the Florida Keys and from nearshore to more than 200 miles offshore.² However, a bloom can remain entirely offshore with minimal effects on the coast. Blooms that affect the Atlantic Coast of Florida and other states farther north (e.g., North Carolina) come from Gulf populations that are entrained into and transported by the Loop Current through the Straits of Florida then into the Gulf Stream proper (Tester et al. 1991; Walsh et al. 2009). Tester et al. (1991) noted a 22–54 day transit time from the Gulf to North Carolina. In some cases, eddies containing cells can break off from the Loop Current and move toward Texas (Steidinger 2009).

The ability of *K. brevis* to survive and out-compete other phytoplankton species has been the subject of much research. The organism is slow-growing and shade-adapted compared to other phytoplankton (e.g., diatoms). Nevertheless, it survives in oligotrophic waters of the West Florida Shelf and can produce high-density, monospecific blooms. In addition to its nutrient physiology and other adaptations described below, the organism could have other strategies that allow it to succeed. For instance, the organism's motility and vertical migration behavior allow it to take advantage of subsurface nutrient pools. Brevetoxins might depress grazing. Allelopathic substances released by *K. brevis* could inhibit competitive species. Evidence of mixotrophy could allow the organism to feed on bacteria and other organic particles to supplement its nutritional requirements.³

Steidinger (2009) suggests that nutrient concentrations in the West Florida Shelf are low despite potentially high estuarine flux and coastal inputs as microscopic and macroscopic inshore plants remove the nutrients effectively in the estuaries before the nutrients can reach 18–74 km offshore; moreover, waters from the Mississippi River plume have lost most of its nutrient load by the time it reaches Tampa Bay. It should be noted that when nutrients become available, Steidinger (2009) proposes that growth rate does not change drastically. Instead, Vargo et al. (2008) suggest that high cell densities might have been transported from other areas or concentrated by physical means (e.g., tides and currents).

One research team has suggested that a significant increase in nutrient inputs to the nearshore waters of West Florida correlates with an increased abundance of *K. brevis* (Brand and Compton

² Heil, C. FWC. January 2010. Personal communication.

³ Heil, C. FWC. January 2010. Personal communication.

2007). Their analysis was based on comparing *K. brevis* abundance to distance off shore. Other researchers have proposed that N fixation with subsequent release of DON and remineralization of nutrients from dead fish can lead to initiation, growth, and maintenance of *K. brevis* blooms on the West Florida and Texas shelves (Walsh et al. 2006; Walsh and Steidinger 2001). In that hypothesis, organic N is provided through N-fixation by the cyanobacterium, *Trichodesmium* spp., which is found to bloom before *Karenia* sp. *Trichodesmium* blooms are thought to be triggered by iron deposition from Saharan dust events. As *Karenia* concentrations increase to fish-killing levels, additional organic nutrients are provided by dead or decaying fish. A recent model, HABSIM, has been developed to project the trajectory of *K. brevis* blooms (Walsh et al. 2009). The model is discussed in more detail in Appendix E. Recent unpublished studies also suggest a shift in the microbial community mediated by toxins or allelopathic substances or both that can alter the nutrient pools.⁴

Researchers computed the N and P flux to sustain *K. brevis* blooms from October 1998 through January 2002. They determined that the nutrient flux from Tampa Bay and Charlotte Harbor could only supply from 5–20 percent and 4–90 percent of the required N and P to meet a moderate bloom (3×10^5 cells L⁻¹) (Vargo et al. 2004; Vargo et al. 2008). They state that most blooms, however, have population levels greater than 3×10^5 cells L⁻¹ and persist for months, but acknowledge that if confined to a small area near the estuary mouth, that the estuarine fluxes could support high populations. Vargo et al. (2008) also note earlier researchers (Froelich et al. 1985; Martin and Kim 1977) correlating P-flux from the Peace River to red tide outbreaks in near coastal waters at Boca Grande, noting that the 1999, 2000, and 2001 red tide blooms had an increased biomass near the estuary mouth and decreased to the north and south, which might be indicative of estuarine efflux contributing to a red tide bloom. That might be the case if nearshore fronts trap blooms, but note that the earlier research was related to a spill of phosphate slime (a byproduct produced by phosphate rock mining).

Two recent reviews (Anderson et al. 2008; Vargo 2009) summarize the current state of the knowledge regarding nitrogen/phosphorus pollution and *K. brevis* blooms in the West Florida Shelf. Vargo (2009) states, “there is no single hypothesis that can account for blooms.” Of approximately 24 thoughts or hypotheses, seven are related to rainfall or riverine flux (or both), six focus on benthic flux in one form or another, seven involve hydrodynamics or are unrelated to the benthos or land sources, and four are based on chemical interactions or allelopathy. Nutrient sources for the growth and maintenance of the blooms include a wide array of sources including atmospheric deposition, N fixation, riverine inputs, benthic fluxes (e.g., from groundwater or remineralized diatom blooms), zooplankton excretion, and even organic matter from the decaying fish killed by *K. brevis* toxins. Anderson et al. (2008) go on to state that “clear evidence to support hypotheses about increased bloom frequency and biomass on the West Florida Shelf is still not yet available.”

B.8.1 Scientific Justification for Not Proposing Harmful Algal Blooms as an Assessment Endpoint

Recent advancements have resulted in what is perhaps the most broadly held working hypothesis for the origination of red tide events—that organic nitrogen is provided through N-fixation by the

⁴ Heil, C. FWC. January 2010. Personal communication.

cyanobacterium, *Trichodesmium* spp., which is found to bloom before *Karenia* sp. *Trichodesmium* blooms are thought to be triggered by iron deposition from Saharan dust events. As *Karenia* concentrations increase to fish-killing levels, additional organic nutrients are provided by dead or decaying fish. Nutrient sources for the growth and maintenance of the blooms include a wide array of sources including atmospheric deposition, N fixation, riverine inputs, benthic fluxes (e.g., from groundwater or remineralized diatom blooms), zooplankton excretion, and even organic matter from the decaying fish killed by *K. brevis* toxins (Vargo 2009). So while it is clear that the red tide is not initiated in response to nutrients from local point and nonpoint sources, there is evidence that nutrient enrichment may prolong blooms once they impinge upon the coast. Nevertheless, current models for the red tide are still at the research level, and thus are not available for use at this time.

B.9 Coral

Coral reefs in Florida face significant degradation due to a number of stressors. These include thermal stress, ocean acidification, tropical storms, increased pollutants from coastal development and runoff, sewage discharge, over-harvesting of grazers and predators, coral diseases, physical damage from ship grounding, coral mining, dredging activities, beach renourishment, decreases in water transparency, invasive species, and damaging fishing practices (Andrews et al. 2005; Collier et al. 2008; Donahue et al. 2008; Szmant 2002). Within the complex Florida reef system other natural factors also affect coral health and distribution, such as internal tidal bores (Leichter et al. 2003), temperature and salinity variability (Porter et al. 1999), grazing of corals and algae, and solar inputs. Combinations of these direct and indirect pressures interact in a complex way to affect coral reef ecology in Florida (Hughes and Connell 1999). Thus, there are multiple factors affecting coral condition and there does not appear to be one main reason behind coral degradation in Florida, although nutrient sources (specifically, the origins of the nitrogen and phosphorus) have been identified as a potential cause.

EPA recognizes that nutrients and water quality play an important role in the viability of coral assemblages. The confounding interaction among multiple and complex stressors that affect overall coral assemblage health makes quantifying nutrient impacts and deriving correspondingly protective nutrient concentrations to support coral health intractable. The discussion below is meant to highlight existing knowledge of the Florida reef system, the state of knowledge surrounding nutrients and coral health, and the interaction of additional stressors that affect coral health.

B.9.1 Corals in Florida

Coral reefs in Florida can be divided into the Florida Reef Tract, Tortugas Banks, and the Southeastern Coast. Most coral are found in shallow waters of less than 18 m in depth, and receive significant influence from the Gulf Stream (Andrews et al. 2005). Coral reefs are known to be highly productive ecosystems but are adapted to nutrient-poor (oligotrophic) conditions.

Major coral reefs in Florida can be characterized as the following (Andrews et al. 2005):

- *Florida Keys Reef Tract*: Extends from Soldier Key to Tortugas Banks, and consist of hard-bottom, patch reef, and bank reef habitat. Hard-bottom habitat is characterized by rock

colonized with calcifying algae, sponges, octocorals, and other species of stony corals. Patch reefs mostly consist of boulder star coral (*Montastraea annularis*). Bank reefs are the farthest from shore and primarily consist of elkhorn coral (*Acropora palmata*).

- *Tortugas Banks*: Developed on karst limestone at depths of 20–40 m, and consist of high coral cover, mostly of the black coral (*Antipatharia*) but have little diversity in species of coral.
- *Southeastern Coast*: Runs northward from northern Monroe County to Martin County in a series of discontinuous reefs, consists of nearshore hard-bottom areas, inner, middle, and outer reefs, each with different characteristics. Nearshore hard-bottom areas consist of macroalgae and numerous small octocorals. Middle reefs consist of octocorals and some species of star corals. The outer reefs consist of more diverse corals including octocorals and barrel sponges (*Xestospongia muta*).

B.9.2 Effects of Nitrogen/Phosphorus Pollution on Coral Reefs

The effect of human inputs to reef systems has been documented on many occasions (Hughes et al. 2003; Knowlton and Jackson 2008; Littler et al. 2009; Mora 2008; Riegl et al. 2009). Many laboratory and field based studies have been conducted to assess the direct and indirect impacts of nitrogen/phosphorus pollution on corals and coral ecosystems (Fabricius 2005). However, results of some of these studies have been controversial when examining the specific role that nutrients play because of varying results, methodologies used, and extraneous influences from multiple sources and stressors (Fabricius 2005; GESAMP 2001; Hughes et al. 1999; Lapointe 1999). Although nitrogen/phosphorus pollution is often considered as one of the factors effecting coral health, it is difficult to clearly quantify the effect that nutrients have on coral health and the concentration that supports a balanced natural population of corals. However, it is important to examine the available literature to determine the range of possible direct and indirect impacts that elevated nutrients can have on corals: results still indicate that there may be some effects from short and long term exposures of coral systems to elevated nitrogen and phosphorus concentrations (Szmant 2002).

B.9.2.1 Direct Impacts

It is difficult to clearly determine the direct physiological impacts of nutrients on corals because results of some of the studies of direct impacts of nutrients on corals have been conflicting or inconclusive (Szmant 2002). Despite the controversy surrounding the results it is important to examine them to determine the range of possible outcomes, as results still indicate that there may be some effect from short and long term exposures of corals to elevated nutrient concentrations (Szmant 2002).

Nutrients may have a direct physiological effect on corals by influencing the coral-zooxanthellae symbiosis and by changing growth, calcification, and reproduction rates (Dubinsky and Stambler 1996; Fabricius 2005). Increases in dissolved inorganic nitrogen (DIN) are more often attributed to changes in zooxanthellae than dissolved inorganic phosphorus (DIP), however changes due to DIP have been observed. The influence of DIN is partly attributed to the idea that zooxanthellae are more often nitrogen limited than phosphorus limited, primarily in areas of high irradiance (Fabricius 2005). Some studies have found that elevated, and particularly imbalanced, nutrients cause an increase in the growth, density, photosynthesis, and respiration rate of N-limited

symbiotic zooxanthellae, which in turn becomes stressful to the coral organism as a whole (Fabricius 2005; Lapointe 2001b). The influence of increased nitrogen on coral production of tissue components such as protein or lipids is mixed (Fabricius 2005). Koop et al. (2001) conducted a field experiment to examine direct effects of additions of DIN or P. They found at high levels of loading that reef organisms and processes were affected by elevated nutrients but that they were neither lethal nor species-specific. They also found that N-fixation increased with phosphate enrichment and denitrification increased with ammonium enrichment. It has been argued by others, such as Szmant (2002), that the results were weak and contradictory, and there is not enough evidence to conclude that nutrients have direct effects on coral reefs. Haas et al. (2009) also conducted a 90-day mesocosm study to examine effect of increased DIN (ammonium, phosphate, nitrate, and all three) and organic nutrients (glucose) on coral–algae assemblages (*Acropora* and typical benthic turf algae) in the Northern Red Sea. They found that increased ammonium concentrations and organic nutrient concentrations stimulate algal growth, while coral tissue pigmentation and chlorophyll *a* content were significantly decreased. In addition, they found that significant changes occurred when algae and increased organic nutrient treatments were combined. The results suggest that enhanced organic nutrients on coral reefs can affect physiology and metabolism of both corals and benthic turf algae. The authors note that “reinforcing interaction between both groups of organisms along with involvement of microbes may facilitate phase shifts in coral reef ecosystems.”

The stress induced by increased nutrients may increase coral’s vulnerability to disease (Szmant 2002). Bruno et al. (2003) reported evidence from field experiments that nutrient enrichment can increase the prevalence of sea fan aspergillosis and yellow band disease in reef-building corals (*Montastraea annularis* and *M. franksii*) in the Florida Keys. Voss and Richardson (2006) found that experimental increases in nutrients resulted in the progression of black band disease and loss in reef framework coral (*Siderastrea sidereal*) in situ and laboratory experiments. Alternately, Patterson et al. (2002) documented populations of *Acropora palmate* being decimated in the Florida Keys by white pox disease, finding that the human fecal bacterium (*Serratia marcescens*) was an agent of the disease. Nugues et al. (2004) also found that physical contact of corals with the macroalga, *Halimeda opuntia*, can trigger white plague type II, suggesting that greater macroalgal cover makes coral reefs more susceptible to disease. Other influences such as temperature, pollution, sedimentation, terrestrial dust input, and runoff have also been implicated in occurrences of coral diseases (Sutherland et al. 2004).

Nutrient stress has been implicated in facilitating coral bleaching events. In general, exposure to a few days of 3–4 degrees Celsius (°C) above normal summer ambient maxima or several weeks of elevations of 1–2°C above maxima can cause coral bleaching. Wooldrige and Done (2009) examined geographic patterns of coral bleaching in 1998 and 2002 on the Great Barrier Reef, Australia, to examine relationships between coral bleaching, heat stress and nutrient flux. They found that corals bathed in nutrient-rich coastal waters have a decreased bleaching resistance (per degree of heating) during the bleaching events compared to reefs in oligotrophic oceanic waters, lowering the upper thermal bleaching threshold by approximately 1.0–1.5°C. Hughes et al. (2003) also examines the role of resiliency of corals and the impact of stressors on corals and notes that nutrient stress is a factor that can increase the susceptibility of corals to bleaching.

B.9.2.2 Indirect Impacts

Many of the effects of nitrogen/phosphorus pollution in coastal and estuarine zones are well-documented (Bricker et al. 2008). Eutrophication is the typical expression of an overabundance of nutrients in estuaries and coasts and is partly defined by an increase in primary production. The increase of primary production changes the physical and biological characteristics of a waterbody. By this mechanism it is possible for the indirect impacts of elevated nutrient concentrations to affect corals. Non-nutrient related impacts also modify the magnitude of indirect impacts of nutrients. For this reason it is difficult to make clear determinations between the secondary effects of nutrient overenrichment and coral health (Fabricius 2005; Hughes et al. 1999; Lapointe 1999). Particularly controversial is the interaction of nutrients, macroalgae, herbivores, and coral. However, there is consistency in some results, and it is important to examine the results of these studies to highlight the possible results of nutrient enrichment on coral even if uncertainty remains.

Nutrient enrichment of the coasts and estuaries has been consistently shown to increase the concentration of phytoplankton in those areas (Bricker et al. 2008). Because phytoplankton decrease the amount of light that reaches the bottom there is less light available for the photosynthetic benthic community. Most corals are necessarily dependent on sunlight and are, thus, typically found in very clear water. When an increase in phytoplankton causes light levels reaching the coral to be reduced (shading), responses include reduced recruitment, decreased calcification, compressed depth distribution zones, low biodiversity at deeper areas, and a shallower depth limit for growth (Lapointe 2001a; Yentsch et al. 2002). These effects were found to be less in shallow water and increased with depth, but responses varied among coral species (Fabricius 2005).

Coral reefs in Florida, especially in southeast Florida, have been documented as having increased numbers of macroalgal blooms and invasions in the past two decades (Lapointe 2007b). However, as demonstrated by the discussion in Section 2.3, the role that nutrients play in macroalgal blooms is a much-debated subject. Factors such as changes in herbivore grazing, storm events, light availability, and temperature changes are often cited influences that interfere with a direct relationship between nutrients and macroalgal abundance (Fabricius 2005; Hughes and Connell 1999; Littler and Littler 2007; Littler et al. 2009; Sotka and Hay 2009). Despite confounding factors of an exact relationship between nutrients and macroalgae blooms, competition between corals and algae has been shown (Lirman and Biber 2000). Macroalgal blooms are often associated with increased shading, outcompeting corals, inhibiting recruitment of juvenile corals, and leading to hypoxia, resulting in detrimental effects on biological diversity (NRC 2000). The transition from a coral to a macroalgal-dominated community can occur through different modes. McCook (1999) conducted a thorough review of existing evidence that nitrogen/phosphorus pollution promote phase shifts from coralline-dominated systems to macroalgal dominated systems. He concluded that nitrogen/phosphorus pollution can contribute to reef degradation but is unlikely to contribute to an immediate phase shift from simply enhancing algal growth rates that would outcompete corals. A combination of other factors in addition to nutrient enhancements, such as reduced herbivory, loss of topographic complexity (from tropical storms), or other disturbances (such as an increase in sedimentation), could

contribute to phase shifts as a whole (discussed more below). A review by McManus and Polsenberg (2004) also came to similar conclusions.

Elevated turbidity associated with increased particulate organic matter (POM) production caused by nitrogen/phosphorus pollution also plays a role in the health of corals (Fabricius 2005). Because corals can use particulate organic matter as a nutrient source, modest increases in particulate organic matter have been found to be beneficial to corals in some instances. However, feeding saturation and ability to use POM varies by species. Excessive levels of POM can prove detrimental to corals causing shading and sedimentation (Rogers 1990). Additionally, the effect of nutrients stimulating increases of particulate organic matter can often be confounded by the fact that land derived inputs to the estuaries and coasts are often in particulate form and often co-occur with eutrophic symptoms.

Increases in nutrient availability can change trophic structures by fueling increased primary production. Increased production in-turn can lead to an increase in the number of herbivores, corallivores, and filter feeders, which could reduce coral cover and speed bioerosion of limestone (Brodie et al. 2005; Fabricius 2005; Lapointe 2001b; Littler et al. 2009; Riegl et al. 2009).

B.9.3 Other Impacts

Often complicating the relationship between nitrogen/phosphorus pollution and coral health are co-occurring stressors that moderate or mimic the effects of nutrients (Littler et al. 2009; Mora 2008). The nutrients that fuel eutrophication are often accompanied by increases in other materials. In many areas excess nutrients are the result of sewage or septic inputs or land derived runoff. Water from these sources often contains increased dissolved and particulate organic material, higher concentrations of non-nutrient pollution, and elevated pathogenic microbial concentration (Dubinsky and Stambler 1996; Fabricius 2005; Mora 2008; Pastorok and Bilyard 1985; Sutherland et al. 2010). Dissolved colored organic materials and particulate loads in water discharged to estuarine and coastal areas decrease the amount of light reaching corals. This shading causes responses like those seen when phytoplankton concentrations increase from increased nutrients and shade corals. Increased particulate loads also fuel sedimentation, a serious problem for corals (Rogers 1990). Chemical pollutants in water discharged to estuaries and coasts can include herbicides, pesticides, pharmaceuticals, a wide variety of household and industrial chemicals, heavy metals, and chemicals that reduce transmission of light to corals. These have all been shown to impact the health of coral communities (Markey et al. 2007; Pastorok and Bilyard 1985). Pathogen loads in runoff and sewage and septic discharges has been implicated in coral disease outbreaks in Florida and other places (Sutherland et al. 2010).

Nutrients other than nitrogen and phosphorus may be influential in reef systems as well. Kline et al. (2006) examined the effects of elevated DOC on coral reefs. The authors suggest that DOC is the cause of coral mortality, not routinely monitored nitrate, phosphate, or ammonia. In addition, they found that enhanced DOC levels accelerate the growth rate of microbes living in the corals' surface mucopolysaccharide layer, which suggests mortality is being caused a disrupted balance between coral and its microbiota. The results show that P and N treatments did not cause significant mortality as opposed to DOC treatments which was approximately five-fold higher (36.6 percent), and significantly different compared to controls ($p < 0.001$).

Increased particulate loads from beach renourishment and dredging can cause shading and sedimentation (Dubinsky and Stambler 1996; Fabricius 2005). Another major anthropogenic impact on coral reefs is related to fishing practices (Mora 2008). If fish populations are harvested in such a way that the trophic balance of the reef environment is upset, then results can be similar to those of eutrophication. Two common examples are the overharvesting of both predators of corals and herbivorous species. Overharvest of predators of coral species can result in an overabundance of coral consumers and lead to higher impacts on corals (Burkepile and Hay 2007). Removal of herbivores has been implicated by several studies as contributing to blooms of macroalgae which can expand in density and range at the expense of corals. As discussed in earlier sections, the correlation between herbivore densities and macroalgal growth is somewhat controversial.

B.9.4 Scientific Justification for Not Proposing Coral as an Assessment Endpoint

The science describing impacts of nitrogen/phosphorus pollution on coral reefs is complex. Corals reef environments are in poor condition globally, including in Florida. Long-term declines in coral condition have been related to multiple causes including thermal stress, ocean acidification, tropical storms, increased pollutants from coastal development and runoff, sewage discharge, over-harvesting of grazers and predators, coral diseases, physical damage from ship grounding, coral mining, dredging activities, beach renourishment, decreases in water transparency, invasive species, and damaging fishing practices (Andrews et al. 2005; Collier et al. 2008; Donahue et al. 2008; Szmant 2002). Experts who have studied the unique south Florida ecosystems cite evidence that increased exposure to nutrients exacerbates the processes or conditions that degrade coral reefs (Bruno et al. 2003; Lapointe 2007b). However, as pointed out by Szmant (2002), it is readily apparent that isolation of nutrient impacts from other stressors would be extremely difficult, if not problematic. Although nitrogen/phosphorus pollution is often considered as one of the factors affecting coral health, it is difficult to clearly quantify the effect that nutrients have on coral health and the concentration that supports a balanced natural population of corals. Overall, the scientific literature indicates that coral populations will benefit generally from decreases in loadings of nutrients to coastal areas.

B.10 Spartina

Spartina spp. seem to be adapted to high nitrogen loads and concentrations (Morris 1991). Addition of nitrogen and ammonia tend to have a greater effect in stimulating growth than urea or phosphorus additions (Buresh et al. 1980; Vince et al. 1981). Standing crops of *Spartina alterniflora* and *S. patens* increased in urea-fertilized plots while the standing crop in phosphate enriched plots resembled control standing crops (Valiela and Teal 1974). Patrick and Delaune (1976) also found *S. alterniflora* to have increased production when exposed to inorganic nitrogen enrichment but no response in production when exposed to phosphorus enrichment. Sullivan and Daiber (1974) also found no response of *Spartina* to increased enrichment of phosphorus. Nitrogen supply seems to be a limiting factor for salt marsh vegetation and *S. alterniflora* (Patrick and Delaune 1976; Valiela and Teal 1974). Broome et al. (1975) found an increase in production of *S. alterniflora* in sandy soils when exposed to nitrogen enrichment alone and then a greater increase in production when phosphorus was also added. Broome et al.

(1975) found no response of *S. alterniflora* to phosphorus enrichment when growing in finer sediments. *Spartina* spp. uptakes phosphorus from sediment and then releases it to the water column (Axelrad et al. 1976) and sedimentation provides the greatest contribution of phosphorus. Addition of nitrogen in mixed stands of *Spartina* may favor another species, such as *Salicornia bigelovii*, and could cause a potential shift of dominant species within the community (Boyer and Zedler 1999; Covin and Zedler 1988). Addition of nitrogen to *Spartina foliosa* did increase production as measured by heights but without continued additions of nitrogen canopy responses were not maintained (Boyer and Zedler 1998). High marsh grasses responded to nutrient enrichment in much the same way as did low marsh grasses (Vince et al. 1981). Low marsh sediments are covered by water at most high tides whereas high marshes are not (Valiela et al. 1978). Freshwater inflows and salinity interactions has an effect on *Spartina* response to nutrient enrichment (Merino et al. 2010). Biomass decreased with increased salinity at four different nutrient concentrations with variation among the nutrient concentrations decreasing as salinity increased (Merino et al. 2010).

B.10.1 Scientific Justification for Not Proposing *Spartina* as an Assessment Endpoint

Spartina was considered as a assessment endpoint because it is a key foundation species (Chanton and Lewis 2002; Verity and Borkman 2009) in some Florida estuaries. EPA has chosen not to consider *Spartina* as an assessment endpoint for numeric criteria development because it is expected to be less sensitive to nutrient enrichment than other endpoints (Verity and Borkman 2009; McFarlin et al. 2008).

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Appendix C. Modeling Tool Overviews

An inventory of the watershed and estuary models that have been previously applied in Florida was developed as part of EPA's analyses. EPA provides a review of these models in this appendix.⁵ The reader is referred to Section 3.3.3 for a description of the models that EPA is considering for use in developing numeric criteria.

⁵ EPA's review of CE-QUAL-ICM, CH3D-IMS, CH3D-SED, CH3D-WES, ECOMSED, EFDC, HSPF, HYCOM, LSPC, SPARROW, WAMView, and WASP is from EPA's 2005 report, *TMDL Model Evaluation and Research Needs*, which is available at <http://www.epa.gov/nrmrl/pubs/600r05149/600r05149.htm>.

C.1 CE-QUAL-ICM Model Description

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Download Information

Availability: Nonproprietary
Cost: N/A

Model Overview/Abstract

The CE-QUAL-ICM water quality model was initially developed as one component of a model package employed to study eutrophication processes in Chesapeake Bay. The ICM/TOXI model is the toxic chemical model and has routines from EPA's WASP (Water Quality Analysis Simulation Program).

Model Features

- Water quality modeling
- Toxics model
- Eutrophication
- Sediment diagenesis model

Model Areas Supported

Watershed	None
Receiving Water	High
Ecological	Low
Air	None
Groundwater	None

Model Capabilities

Conceptual Basis

There are two distinctly different development pathways to ICM: a eutrophication model (ICM) and an organic chemical model (ICM/TOXI). The model employs an unstructured grid system, which facilitates linkage to a variety of hydrodynamic models.

Scientific Detail

- ICM stands for "integrated compartment model," which is analogous to the finite-volume numerical method. The model computes constituent concentrations resulting from transport and transformations in well-mixed cells that can be arranged in arbitrary one-, two-, or three-dimensional configurations. Thus, the model employs an unstructured grid system.
- The release version of the eutrophication model computes 22 state variables, including physical properties; multiple forms of algae, carbon, nitrogen, phosphorus, and silica; and dissolved oxygen. Recently, two size classes of zooplankton, two benthos compartments (deposit feeders and filter feeders), submerged aquatic vegetation (roots and shoots biomass), epiphytes, and benthic algae were added, although this version of the code is not generally released to the public.
- Each state variable may be individually activated or deactivated.
- One significant feature of ICM, eutrophication version, is a diagenetic sediment submodel. The sub-model interactively predicts sediment-water oxygen and nutrient fluxes. Alternatively, these fluxes may be specified based on observations.
- The ICM/TOXI model resulted from incorporating the toxic chemical routines from EPA's WASP (Water Quality Analysis Simulation Program) model into the transport code for ICM, incorporating a more detailed benthic sediment model, and enhancing linkages to sediment transport models.
- ICM/TOXI includes physical processes, such as sorption to DOC and three solid classes, volatilization, and sedimentation. It also includes chemical processes such as ionization, hydrolysis, photolysis, oxidation, and biodegradation.
- ICM/TOXI can simulate temperature, salinity, three solids classes, and three chemicals (total chemical for organic chemicals and trace metals). Each species can exist in five phases (water, DOC-sorbed, and sorbed to three solids types) via local equilibrium partitioning.

Model Framework

The model consists of a main program, an INCLUDE file, and subroutines. Both the main program and subroutines perform read and write operations on numerous input and output files. The model does not compute hydrodynamics. Flows, diffusion coefficients, and volumes must be specified externally and read into the model. Hydrodynamics may be specified in binary or ASCII format and may be obtained from a hydrodynamics model such as the CH3D-WES or EFDC.

Scale

Spatial Scale

- One-, two-, or three-dimensional

Temporal Scale

- User-defined timestep. The timestep may be varied through the auto-stepping option or at discrete, user-specified intervals.

Assumptions

The model assumes that the dynamics of each physical, chemical, and biological component can be described by the principle of conservation of mass.

Model Strengths

- The model has a predictive diagenetic sediment submodel that interactively predicts sediment-water oxygen and nutrient fluxes.
- The ICM/TOXI model has toxic chemical routines, which further enhance linkage with sediment transport models.

Model Limitations

- The model does not compute hydrodynamics. Flows, diffusion coefficients, and volumes must be specified externally and read into the model.
- The user must provide processors, which prepare input files and process output for interpretation and presentation.

Application History

The ICM eutrophication model has been applied to a variety of sites, including Chesapeake Bay, Inland Bays of Delaware, New York Bight, Newark Bay, New York–New Jersey Harbors and Estuaries, Lower Green Bay, Los Angeles–Long Beach Harbors, Cache River wetland, San Juan Bay and Estuaries, Florida Bay, and Lower St. Johns River (on-going).

The WASP toxic chemical model on which ICM/TOXI is based has been applied to a wide variety of sites. CE-QUAL-ICM also has been linked to the CH3D-WES and EFDC hydrodynamic models.

Model Evaluation

Not available

Model Inputs

- Initial conditions
- Time sequences of boundary conditions (inputs from watershed sources and discharges)
- Reservoir geometry
- Physical coefficients
- Biological and chemical reaction rates
- Time sequences of meteorological data used to compute temperature.

Users' Guide

Available online: <http://www.wes.army.mil/el/elpubs/pdf/trel95-15.pdf>

Technical Hardware/Software Requirements

Computer hardware:

- PC

Operating system:

- PC-DOS. Operates on a variety of platforms, including 486 PC, Silicon Graphics and Hewlett Packard workstations, and Cray Y-MP and C-90 mainframes

Programming language:

- FORTRAN 77

Runtime estimates:

- Minutes to hours

Linkages Supported

CH3D-WES, EFDC

Related Systems

Surface Water Modeling System (SMS)

Sensitivity/Uncertainty/Calibration

Highly accurate for simulation of reservoir systems with adequate monitoring data and application experience.

Model Interface Capabilities

Not available

References

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C.2 CH3D-IMS: Curvilinear-grid Hydrodynamics 3D—Integrated Modeling System Model Description

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Download Information

Availability: Proprietary
Cost: Contact Dr. Y. Peter Sheng at pete@coastal.ufl.edu

Model Overview/Abstract

CH3D-IMS is an integrated modeling system based on a CH3D model framework. It models circulation, wave, sediment transport, water quality, light attenuation, and seagrass on curvilinear grids. The circulation is solved using CH3D. Wave is modeled using SWAN framework. Four additional modules: CH3D-SED3D, CH3D-WQ3D, CH3D-LA, and CH3D-SAV are used for calculating the sediment transport, water quality, light attenuation, and seagrass.

Model Features

- Three-dimensional hydrodynamics
- Cohesive and noncohesive sediment transport
- Nitrogen, phosphorus, phytoplankton, zooplankton, and dissolved oxygen
- Light attenuation and seagrass kinetics

Model Areas Supported

Watershed	None
Receiving Water	High
Ecological	High
Air	None
Groundwater	None

Model Capabilities

Conceptual Basis

The waterbody is conceptualized as a series of grid points on a curvilinear orthogonal coordinate system.

Scientific Detail

The details of the circulation model are presented in the fact sheet of CH3D. The governing equations for sediment transport and water quality are solved on a nonorthogonal curvilinear coordinate on the horizontal plane. In the vertical direction, both the σ -coordinate and z-

coordinate are provided. Sediment transport and water quality are solved using the same timestep as the hydrodynamic calculation. The sediment transport processes include advection, turbulent mixing, settling/flocculation, deposition, and resuspension. Wave-current interaction inside the bottom boundary layer also is considered. For water quality simulation, the nitrogen cycling models dissolved, particulate, organic, and inorganic nitrogen species; the phosphorus cycling models dissolved, particulate, organic, and inorganic phosphorus species. In addition, dissolved oxygen, phytoplankton, and zooplankton are modeled. The seagrass module calculates the growth and decay of seagrass biomass due to light, nutrient, temperature, and salinity.

Model Framework

- Three-dimensional model
- River, lake, reservoir, estuary, ocean

Scale

Spatial Scale

- One-dimensional, two-dimensional, and three-dimensional

Temporal Scale

- User-defined timestep

Assumptions

- Hydrostatic assumption
- Boussinesq approximation
- Reynold's stress assumption

Model Strengths

- Capable of modeling one-dimensional, two-dimensional, and three-dimensional hydrodynamics, sediment transport, and eutrophication in various waterbodies with complex bathymetry.
- Boundary-fit curvilinear coordinate can represent the waterbody boundaries accurately.
- The σ and z coordinates in the vertical direction provides flexible options for modeling waterbodies with different bathymetry.

Model Limitations

- Not a public modeling system.
- No source code available.
- No user's manual.

Application History

The model has been applied in Chesapeake Bay, James River, Lake Okeechobee, Sarasota Bay, Tampa Bay, Indian River Lagoon, Florida Bay, St. Johns River, Biscayne Bay, Charlotte Harbor, Gulf of Mexico, and Pinellas County and offshore.

Model Evaluation

The model has been evaluated in many journal and conference papers.

Model Inputs

- Initial conditions
- Time sequences of boundary conditions (inputs from watershed sources and discharges)
- Reservoir geometry
- Physical coefficients
- Biological and chemical reaction rates
- Time sequences of hydrometeorological conditions

Users' Guide

Not available

Technical Hardware/Software Requirements

Computer hardware:

- VAX, SGI, SUN, DEC, IBM, IBM-PC

Operating system:

- PC-DOS, UNIX, WINDOWS, Linux

Programming language:

- FORTRAN

Runtime estimates:

- Minutes to hours

Linkages Supported

None

Related Systems

POM, EFDC, ECOMSED, GLLVHT

Sensitivity/Uncertainty/Calibration

Not available

Model Interface Capabilities

- ArcView-based GUI for grid generation, pre- and post- processing

References

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C.3 CH3D-SED (& CH3D-WES): Curvilinear Hydrodynamics in Three Dimensions Model Description

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Download Information

Availability: Available only to Department of Defense Agencies
Cost: N/A

Model Overview/Abstract

CH3D-SED is the newly developed mobile bed version of CH3D-WES, which was developed for the Chesapeake Bay Program. The USACE is using it to investigate sedimentation on bendways, crossings, and distributaries on the lower Mississippi and Atchafalaya rivers. These applications address dredging, channel evolution, and channel training structure evaluations. CH3D-SED functions as a hydrodynamic (through the incorporation of CH3D-WES) and sediment transport model. Physical processes affecting circulation and vertical mixing that can be modeled include tides, wind, density effects (salinity and temperature), freshwater inflows, turbulence, and the effect of the earth's rotation. CH3D-SED can be applied to rivers, lakes, reservoirs, estuaries, or coastal waters.

Model Features

- Hydrodynamic
- Sediment transport
- Linkage to CE-QUAL-IC water quality model

Model Areas Supported

Watershed	None
Receiving Water	High
Ecological	None
Air	None
Groundwater	None

Model Capabilities

Conceptual Basis

The CH3D-SED hydrodynamic and sediment transport model is based on an extension of the stretched vertical coordinate version of the CH3D-SED by Spasojevic and Holly (1997) to

include cohesive sediment transport. The model is capable of two- or three-dimensional operation and employs standard formulations for settling, deposition, and resuspension.

CH3D-WES makes hydrodynamic computations on a curvilinear or boundary-fitted platform grid. Deep navigation channels and irregular shorelines can be modeled because of the boundary-fitted coordinates feature of the model. Vertical turbulence is predicted by the model and is crucial to a successful simulation of stratification, destratification, and anoxia. A second-order model based on the assumption of local equilibrium of turbulence is employed.

A boundary-fitted, nonorthogonal, finite-difference approximation in the horizontal plane and a sigma-stretched approximation in the vertical direction are used for the approximations of the governing equations.

Scientific Detail

The hydrodynamic model solves the depth-averaged Reynolds approximation of the momentum equation for velocity and the depth-averaged conservation of mass equation for water surface elevation. The three-dimensional velocity field is determined by computing the deviation from the depth-averaged velocity by solving the conservation of mass equation in conjunction with a k-e closure for vertical momentum diffusion.

Sedimentation computations are based on a two-dimensional solution of the conservation of mass equation for the channel bed, and three-dimensional advection-diffusion equation for suspended sediment transport. The sediment transport algorithms independently account for the movement of sediment as either bed load or suspended load, as well as the exchange of sediment between these two modes of transport. The model is also generalized for application to mixed-grain-size sediments, with appropriate bed material sorting and armoring routines. The formulation to a user-specified multiple-grain-size distribution uniquely allows the simulation of erosion, entrainment, transport, and deposition of contaminated sediments on the bed and in the water column. A contaminated sediment associated with a given grain size can be independently accounted for by applying a small dimensional perturbation from the reference grain size. This perturbation has negligible effects on sediment mobility characteristics. Because each grain size specification is independently tracked, however, tracking of zones of contaminated bed material is possible.

Model Framework

- Curvilinear Finite Difference Numerical Formulations for Hydrodynamic and Sediment Transport

Scale

Spatial Scale

- Three-dimensional

Temporal Scale

- Dynamic

Assumptions

- Based on accepted formulations of the three-dimensional, hydrostatic, hydrodynamic equations and conservative transport equation.

Model Strengths

- Strong capabilities for hydrodynamics and sediment transport.

Model Limitations

- Considerable technical expertise in hydrodynamics is required to use the model effectively.

Application History

CH3D-SED was applied to investigate maintenance dredging quantities for channel alignment studies on the lower Atchafalaya River (Hall 1996). The model successfully reproduced existing sediment deposition quantities and locations, and it was instrumental in the decision made by the local sponsor to maintain the existing channel alignment.

The capabilities of CH3D are illustrated by its application to the Chesapeake Bay. The numerical grid employed in the Chesapeake Bay model has 734 active horizontal cells and a maximum of 15 vertical layers, resulting in 3,992 computational cells. Grid resolution is 1.52 m vertical and approximately 10 km longitudinal and 3 km lateral. The x, y coordinates of the grid are transformed into the ζ -curvilinear coordinates to allow for better handling of the complex horizontal geometries. Velocity is also transformed so that its components are perpendicular to the ζ -coordinate lines, thus allowing boundary conditions to be prescribed on a boundary-fitted grid in the same manner as on a Cartesian grid. Major tributaries are modeled three-dimensionally in the lower reach of the bay and two-dimensionally with constant width in the upper reach.

Cerco et al. (1993) used CH3D-WES in conjunction with CE-QUAL-ICM to predict water column and sediment processes that affect water quality in the Chesapeake Bay. Data from 1984–1986 were used, and the linked modeling approach was successful in predicting the spring algal bloom, onset and breakup of summer anoxia, and coupling of organic particle deposition with sediment-water nutrient and oxygen fluxes.

Model Evaluation

Johnson et al. (1993) validated the model by applying it to six datasets. The first three datasets contained approximately one month's worth of data each and represented a dry summer condition, a spring runoff, and a fall wind-mixing event. The last three applications were yearlong simulations for 1984 (a wet year), 1985 (a dry year), and 1986 (an average year). Results demonstrate that the model is a good representation of the hydrodynamics of the Chesapeake Bay and its major tributaries.

Model Inputs

- Time-varying water-surface elevations, salinity, and temperature conditions at the ocean entrance and at freshwater inflows at the head of all tributaries.
- Time-varying wind and surface heat exchange data at one or more locations.

- All input data, including initial conditions, bathymetry, boundary, and computational control data are input from fixed files.

Users' Guide

Not available

Technical Hardware/Software Requirements

Computer hardware:

- Unix workstation or super computer.

Operating system:

- Unix

Programming language:

- FORTRAN

Run time estimates:

- Compute intensive
- Run time is highly dependent on computer hardware, model domain spatial resolution, the period of prototype conditions simulated and other options such as whether the model is simulation only hydrodynamic or hydrodynamics and the fate and transport of dissolved and suspended material. Under this wide range of variability, simulations could require minutes to weeks.

Linkages Supported

CE-QUAL-ICM

Related Systems

Surface Water Modeling System (SMS)

Sensitivity/Uncertainty/Calibration

Not available

Model Interface Capabilities

- US Army Corps of Engineers Surface Water Modeling System (SMS)

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C.4 DYNHYD Model Description

DYNHYD5 solves the one-dimensional equations of continuity and momentum conservation for a branching or channel-junction (link-node) computational network. Driven by variable upstream flows and downstream heads, simulations typically proceed at 1- to 5-minute intervals. The resulting unsteady hydrodynamics are averaged over larger time intervals and stored for later use by WASP5.

The hydrodynamic model solves one-dimensional equations describing the propagation of a long wave through a shallow water system while conserving both momentum (energy) and volume (mass). The equation of motion, based on the conservation of momentum, predicts water velocities and flows. The equation of continuity, based on the conservation of volume, predicts water heights (heads) and volumes. This approach assumes that flow is predominantly one-dimensional, that Coriolis and other accelerations normal to the direction of flow are negligible, that channels can be adequately represented by a constant top width with a variable hydraulic depth (i.e. "rectangular"), that the wave length is significantly greater than the depth, and that bottom slopes are moderate. Although no strict criteria are available for the latter two assumptions, most natural flow conditions in large rivers and estuaries would be acceptable. Dam-break situations could not be simulated with DYNHYD5, nor could small mountain streams. The DYNHYD model can be downloaded at <http://eng.odu.edu/cee/resources/model/wasp.shtml>.

C.5 ECOMSED: Estuary and Coastal Ocean Model with Sediment Transport Model Description

Contact Information

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http://www.hydroqual.com/ehst_ecomsed.html

Download Information

Availability: Nonproprietary Version Available
Register to download files: http://www.hydroqual.com/ehst_ecomsed.html
Cost: N/A

Model Overview/Abstract

ECOMSED is a three-dimensional hydrodynamic and sediment transport model. The hydrodynamic module solves the conservation of mass and momentum equations with a 2.5-level turbulent closure scheme on a curvilinear orthogonal grid in horizontal plane and σ -coordinate in the vertical direction. Water circulation, salinity, and temperature are obtained from the hydrodynamic module. The sediment transport module computes the sediment settling and resuspension processes for both cohesive and noncohesive sediments under the impact of waves and currents. The hydrodynamic component is same as the ECOM3D/POM model.

Model Features

- Three-dimensional hydrodynamics
- Cohesive and noncohesive sediment transport
- Sediment-bound and dissolved tracer transport
- Wind-waves-generated shear stress

Model Areas Supported

Watershed	None
Receiving Water	High
Ecological	None
Air	None
Groundwater	None

Model Capabilities

Conceptual Basis

The waterbody is conceptualized as a series of grid points on a curvilinear orthogonal coordinate system.

Scientific Detail

The governing equations of the hydrodynamic component in ECOMSED are the continuity equation, Reynold's equations, heat and salinity transport equations on curvilinear-orthogonal grid on the horizontal plane and σ -coordinate in the vertical direction. It uses a 2.5-level turbulence closure scheme that solves the transport of turbulent kinetic energy and turbulent macroscale. The governing equation of sediment transport is an advection-dispersion equation that uses the hydrodynamic results. The hydrodynamic governing equations are solved using a mode-splitting technique. The external mode that contains fast moving gravity wave is solved with small timesteps to ensure stability, whereas the internal mode uses large timesteps to save the computation time. Finite difference of the differential equation is applied on a staggered C grid in space, and the three-time-level leap-frog scheme is applied for the timestepping. Three schemes, including central difference, upwind difference, and the multidimensional positive definite advection scheme are provided in the model to solve the advection term in the transport equations. The sediment transport component uses the same grid, structure, and computational framework as the hydrodynamic component to simulate the settling, deposition, and resuspension of both cohesive and noncohesive sediments. The Grant-Madson wave-current model is incorporated in ECOMSED to account for wind-wave-generated shear stress.

Model Framework

- Three-dimensional model
- River, lake, reservoir, estuary, ocean

Scale

Spatial Scale

- One-, two-, and three-dimensional

Temporal Scale

- User-defined timestep

Assumptions

- Hydrostatic assumption
- Boussinesq approximation
- Reynold's stress assumption

Model Strengths

- ECOMSED is capable of modeling one-, two-, and three-dimensional hydrodynamics in various water bodies with complex bathymetry.
- The boundary-fit curvilinear coordinate can represent the waterbody boundaries accurately with fewer grids than Cartesian coordinate.
- The vertical σ -coordinate represents the bathymetry without assuming rectangular bottom boundary.

Model Limitations

- The vertical σ -coordinate may cause significant pressure gradient error at areas with sharp bottom elevation change.

- Timestep and grid size need to be chosen carefully to balance the computation time and model resolution and ensure model stability.

Application History

ECOMSED has been applied to Chesapeake Bay, New York Bight, Delaware Bay, Delaware River, Gulf Stream Region, Massachusetts Bay, Georges Bank, the Oregon Continental Shelf, New York Harbor, and Onondaga Lake.

Model Evaluation

The hydrodynamic component of ECOMSED is based on Princeton Ocean Model, which has been tested and applied by various users. The theory and model testing history can be found in journal and conference papers.

Model Inputs

- Initial conditions
- Bathymetry and waterbody boundaries
- Physical coefficients
- Water surface elevations or flow rate at open boundary
- Time sequences of hydrometeorological conditions

Users' Guide

Available online (after registering): http://www.hydroqual.com/ehst_ecomsed.html

Technical Hardware/Software Requirements

Computer hardware:

- PC, workstation, and mainframe

Operating system:

- PC-DOS, Unix, Windows

Programming language:

- FORTRAN

Runtime estimates:

- Minutes to hours

Linkages Supported

Linked with HydroQual's RCA model.

Related Systems

POM, EFDC, CH3D, GLLVHT

Sensitivity/Uncertainty/Calibration

Not available

Model Interface Capabilities

Input file in text format.

References

HydroQual, Inc. 2002. *A Primer for ECOMSED version 1.3*. (Computer program manual).

HydroQual, Inc., Mahwah, NJ.

C.6 EFDC: Environmental Fluid Dynamics Code Model Description

Contact Information

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Download Information

Availability: Nonproprietary
Cost: N/A

Model Overview/Abstract

The EFDC model is single-source-code three-dimensional modeling system having hydrodynamic, water quality-eutrophication, sediment transport, and toxic contaminant transport components transparently linked together. The model can execute in a fully coupled mode, simultaneously simulating hydrodynamics and sediment and contaminant transport, or in a transport-only mode, using saved hydrodynamic transport information. The EFDC model uses a finite difference spatial representation and is capable of reduced dimension execution in one-dimensional network and two-dimensional (horizontal or vertical plane) modes. Water column transport includes three-dimensional advection and vertical and horizontal turbulent diffusion. Shear dispersion may be included for two-dimensional horizontal applications. A water quality-eutrophication model, functionally equivalent to CE-QUAL-IC, is also incorporated into EFDC (Hamrick and Wu, 1997; Park, et al., 1995). The model can be applied to rivers, lakes, reservoirs, estuaries, wetlands, and coastal regions.

Model Features

- General purpose three-dimensional hydrodynamic and transport model
- Model simulates tidal, density, and wind-driven flow; salinity; temperature; and sediment transport
- Two built-in, fully coupled water quality/eutrophication submodels are included in the code, as well as a toxicant transport and fate model

Model Areas Supported

Watershed	Low
Receiving Water	High
Ecological	High

Air	None
Groundwater	Low

Model Capabilities

Conceptual Basis

The EFDC model solves the vertically hydrostatic, free-surface, variable-density, turbulent-averaged equations of motion and transport equations for turbulence intensity and length scale, salinity, and temperature in a stretched, vertical coordinate system and in horizontal coordinate systems that may be Cartesian or curvilinear-orthogonal. Equations describing the transport of suspended sediment, toxic contaminants, and water quality state variables are also solved. Multiple size classes of cohesive and noncohesive sediments and associated deposition and resuspension processes and bed geomechanics are simulated. Toxics are transported in both the water and sediment phases in the water column and bed. The built-in 21-state-variable water quality model is based on the CE-QUAL-ICM reaction kinetic. A 10-state-variable reduced water quality model is functionally equivalent to WASP5. Other model features include drying and wetting, hydraulic structure representation, vegetation resistance, and Lagrangian particle tracking. The model also accepts radiation stress fields from wave refraction-diffraction models, which allows simulation of longshore currents and sediment transport.

Scientific Detail

The EFDC model framework includes methods for computing hydrodynamics, mixing zone dilution, eutrophication, sediment transport, and toxic contaminant transport.

Mixing Zone

A Lagrangian buoyant jet near-field dilution and mixing-zone model is embedded within the far field solution allowing representation of the local distribution of contaminated sediment near point sources.

Hydrodynamic

EFDC uses a finite difference scheme with three time levels and an internal-external mode splitting procedure to achieve separation of the internal shear or baroclinic mode from the external free-surface gravity wave or barotropic mode. An implicit external mode solution is used with simultaneous computation of a two-dimensional surface elevation field by a multicolor successive overrelaxation procedure. The external solution is completed by calculation of the depth-integrated barotropic velocities using the new surface elevation field. Various options can be used for advective transport in EFDC. These include the “centered in time and space” scheme, and the “forward in time and upwind in space” scheme.

Sediment Transport

The sediment transport component simulates a user specified number of size classes of cohesive and noncohesive sediment. Sediment settling is represented by concentration and ambient-flow-turbulence-dependent formulations to represent hindered settling of noncohesive sediment and approximately represent aggregation and disaggregation of cohesive sediment. Water column-bed sediment and sorbed contaminant exchange is

represented by deposition and erosion fluxes. For noncohesive sediment, the net flux is represented as dependent on the bed stress, the near bottom and bed surface sediment concentration, and the critical Shield's parameter. For cohesive sediment, deposition and erosion fluxes are dependent on the bed stress, critical deposition and erosion stresses, and the shear strength of the bed. The sediment bed is represented by a time varying number of layers. Sediment in each layer is characterized by mass per unit area, void ratio, and shear strength. The void ratio, of the layers is specified or determined by a bed consolidation model with shear strength being determined as a function of void ratio.

Contaminant Transport

Vertical transport of sediment and sorbed contaminants between bed layers is implicitly represented by sediment particle displacement in response to layer thickness variations dynamically determined by the consolidation model. Transport of dissolved contaminants between the water column and bed and between bed layers includes pore water advection, dynamically determined by the bed consolidation model and pore water diffusion. An arbitrary number of toxic contaminants can be simultaneously transported. The simple contaminant processes option includes constant coefficient equilibrium partitioning, volatilization, and lumped first-order decay with unique coefficients for the water column and sediment bed. The complex contaminant processes option allows for solids-concentration-dependent partitioning and specification of ambient-environment-dependent volatilization, hydrolysis, photolysis, oxidation, and biodegradation reactions specific to the contaminants being simulated.

Ecosystem

The model is based on the CE-QUAL-ICM and incorporates a predictive sediment process or diagenesis model (DiToro and Fitzpatrick, 1993). The eutrophication model is directly coupled to the hydrodynamic model and is capable of two and three-dimensional spatial resolution. Water column state variables include up to three algae classes represented in carbon equivalent units, ammonia, nitrite-nitrate, organic nitrogen, orthophosphate or inorganic phosphorous, organic phosphorus, organic carbon, chemical oxygen demand, dissolved oxygen, available and unavailable silica, and total active metal, which is used as a sorption site. Organic carbon, nitrogen, and phosphorous are subdivided into three classes: dissolved, labile particulate, and refractory particulate. Model variables in the sediment bed include particulate organic carbon, nitrogen, and phosphorous, each in three reaction rate classes; particulate and available silica; sulfide or methane; ammonia; nitrate; inorganic phosphorus; bed-water column fluxes of ammonia, nitrate, inorganic phosphorous and silica; sediment oxygen demand; and release of chemical oxygen demand. The model's formulation allows direct determination of organic carbon levels in the water column and sediment bed.

Model Framework

- Three-dimensional curvilinear-orthogonal finite difference

Scale

Spatial Scale

- Three-dimensional

Temporal Scale

- Dynamic

Assumptions

- Based on accepted formulations of the three-dimensional hydrostatic hydrodynamic equations and conservative transport equation

Model Strengths

- Completely integrated three-dimensional hydrodynamics, water quality/eutrophication, and sediment-contaminant transport

Model Limitations

- Requires considerable technical expertise in hydrodynamics to use the model effectively
- Requires expertise in eutrophication processes to use the water quality component

Application History

The EFDC model has been used for modeling studies in the estuaries of the Chesapeake Bay System, the Indian River Lagoon and Lake Okeechobee in Florida, the Peconic Bay System in New York, Stephens Passage in Alaska, and Nan Wan Bay in Taiwan. The model has also been used to simulate large-scale wetlands flow and transport in the Everglades. The EFDC model has been applied extensively for circulation, discharge dilution, and water quality/eutrophication studies (Hamrick, 1992b; Tetra Tech, 1994, 1995, 1998). The model has also been applied for estuarine-cohesive sediment transport simulation (Yang, 1996), and coastal noncohesive sediment transport (Zarillo and Surak, 1995), and heavy metals and organic contaminant transport (Schock and Hamrick, 1998).

Model Evaluation

- Not available

Model Inputs

- Open boundary water surface elevation
- Wind and atmospheric thermodynamic conditions
- Open boundary salinity and temperature
- Volumetric inflows
- Inflowing concentrations of sediment and water quality state variables
- Input file templates are included with the source code and the user's manual to aid in input data preparation

Users' Guide

- Contact the developer at john.hamrick@tetrattech-ffx.com.

Technical Hardware/Software Requirements

Computer hardware:

- PC, MacIntosh, Unix workstation, Super computer

Operating system:

- Windows, Mac OS, Unix, Linux

Programming language:

- FORTRAN

Runtime estimates:

- Can be computation intensive but is highly optimized and has faster documented runtime than similar software systems
- Runtime is highly dependent on computer hardware, model domain spatial resolution, the period of prototype conditions simulated, and other options, such as whether the model is simulation-only hydrodynamic or hydrodynamics and the fate and transport of dissolved and suspended material; under this wide range of variability, simulations could require minutes to weeks

Linkages Supported

- Linkages to WASP, CE-QUAL-ICM, RCA, and a generic output for food chain and risk assessment model

Related Systems

GridEFDC, EFDCexplorer, EFDCview, EPA TMDL Toolbox

Sensitivity/Uncertainty/Calibration

Not available

Model Interface Capabilities

- Public domain GUI including EFDCexplorer and EFDCview

References

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C.7 HABSIM Model Description

The HABSIM model is a coastal ocean model modeling system developed at the University of South Florida which is comprised of biological, chemical, atmospheric, benthic and physical sub-models specifically designed to simulate harmful algae blooms including those in the Gulf of Mexico. The biological component includes multiple classes of plankton, zooplankton, bacteria and fish. The chemical model represents the marine carbon, nitrogen, phosphorous, silica and iron cycles. The benthic model represents benthic diatom and nutrient regeneration. The biological, chemical, and benthic components are presumed to be dynamically coupled while the atmospheric sub-model, which provides iron deposition, and the physical circulation model which provides advective and diffusion transport and salinity and temperature, are essentially external with no feedback from the biological, chemical and benthic components.

The structure of the biological, chemical, and benthic components is similar to current state of the art coastal and estuarine eutrophication models such as CE-QUAL-ICM, EFDC, and WASP which represent multiple plankton species and the carbon, nitrogen, and phosphorous cycles. These models differ from HABSIM in that they include settling of particulate organic C, N, and P classes which provides organic material to sediment diagenesis sub-models which predict sediment oxygen demand and nutrient fluxes back into the water column. Based on the limited material available, the HABSIM benthic sub-model does not have a full diagenesis model. Applications of coastal and estuarine eutrophication models generally focus on land source nutrient enrichment leading to non-toxic algae blooms and resulting hypoxic conditions. However, HABSIM is more specifically focused on simulation of harmful algae, including *K. brevis*, resulting from land, offshore, and atmospheric nutrient sources.

A suite of circulation or hydrodynamic models can provide the functionality for the HABSIM physical sub-model. There are two ROMS (Regional Ocean Models Systems) model configurations, a west Florida shelf configuration of FVCOM (Finite Volume Coastal Ocean Model), and another larger domain configuration for the south Atlantic Bight and Gulf of Mexico (SABGOM). Open boundary conditions for the two ROMS models are provided by the global scale HYCOM (Hybrid Coordinate Ocean Model) model. The open boundary conditions for the FVCOM model appear to be provided by west Florida Shelf ROMS model, but could also be provided by the SABGOM ROMS model or the HYCOM model. The procedure of using a larger spatial coverage model to provide boundary conditions for a small cover, high resolution model is referred to as nesting and more specifically one-way nesting in these cases. The exact details of whether the ROMS and FVCOM models externally provide physical transport to the HABSIM biochemical sub-models or whether these models are hosted within ROMS and FVCOM is somewhat unclear, however the external linkage appears most likely.

The HYCOM model is considered one of the most advanced global scale ocean circulation models. Originally developed at the University of Miami, continued development occurs there and at a number of federal laboratories including the Naval Research Laboratory. Previous regional versions specific to the Gulf of Mexico and the North Atlantic have been superseded by the 1/12 degree global configuration at the Naval Research Laboratory. HYCOM is computationally intensive and requires high performance computing resources such as those at NRL and other government laboratories and a few universities. Its complex hybrid vertical grid

system and structured pseudo-Cartesian longitude-latitude grid system make it impractical for shallow water applications including estuaries and bays. Conversely, HYCOM provides an excellent source of boundary conditions for limited area shelf models.

The ROMS model has been applied extensively at a range of scales, from large estuaries such as the Chesapeake Bay to regional or sub-basin ocean scales such as the west Florida shelf and SABGOM configurations used as the physical component of HABSIM. The major computational components of ROMS were developed at Rutgers University and UCLA. ROMS uses a generalized stretched bottom and free surface following vertical coordinate combined with a structured curvilinear-orthogonal horizontal coordinate system making it suitable for shallow water applications. The ROMS model can operate on desktop workstations and small clusters and thus has a wide user base and peer reviewed literature documenting its application. There are various references in the gray literature on developing internally coupled biogeochemical components for ROMS although none have been reported in the literature.

The FVCOM model is a relatively new coastal and estuarine circulation and transport model developed at the University of Massachusetts, Dartmouth. FVCOM utilizes a generalized stretched bottom and free surface following vertical coordinate in combination with an unstructured triangular grid representation in the horizontal. The unstructured triangular horizontal grid is advantageous for allowing larger variations in grid scale in a single configuration and greater flexibility in resolving complex shorelines than quadrilateral structured grid formulations. The finite volume formulation resolves many of the conservation and oscillation problems associated with finite element based unstructured grid models. FVCOM has a growing research and academic user base and peer reviewed literature.

Overall the HABSIM modeling system appears to be a very promising tool for studying and ultimately predicting harmful algal blooms. The physical or circulation model components ROMS or FVCOM are considered operational models while the biological, chemical, and benthic sub-models are still at a research level, as extensive documentation of the formulations and applications including peer reviewed publications are not currently available. However, these model components certainly have the potential to reach operational status in the near future. The lack of a number of features in the biological and chemical components currently limits their applicability to general estuarine and coastal eutrophication applications, although these empirical formulations can be added to the model.

C.8 HSPF: Hydrologic Simulation Program FORTRAN Model Description

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AQUA TERRA

<http://www.aquaterra.com/>

Download Information

Availability: Nonproprietary

Cost: N/A

Model Overview/Abstract

HSPF is a comprehensive package for simulating watershed hydrology and water quality for a wide range of conventional and toxic organic pollutants. With its predecessors dating back to the 1960s, HSPF is the culminating evolution of the Stanford Watershed Model (SWM), watershed-scale Agricultural Runoff Model (ARM), and Nonpoint Source Loading Model (NPS) into an integrated basin-scale model that combines watershed processes with in-stream fate and transport in one-dimensional stream channels. HSPF simulates watershed hydrology, land and soil contaminant runoff, and sediment-chemical interactions. The model can generate time series results of any of the simulated processes. Overland sediment may be divided into three types of sediment (sand, silt, and clay) for in-stream fate and transport. Pollutants interact with suspended and bed sediment through soil-water partitioning. The most recent release is HSPF Version 12, which is distributed as part of the EPA BASINS (Better Assessment Science Integrating Point and Nonpoint Sources) system.

Model Features

- Detailed watershed simulation model
- Watershed hydrology
- Runoff/sediment/pollutant generation and transport
- One-dimensional stream hydrology and transport
- Pesticide fate and transport simulation

Model Areas Supported

Watershed	High
Receiving Water	Medium
Ecological	None
Air	None
Groundwater	Low

Model Capabilities

Conceptual Basis

In HSPF, a subwatershed is typically conceptualized as a group of various land uses all routed to a representative stream segment. Several small subwatersheds and representative streams may be networked together to represent a larger watershed drainage area. Various modules are available and may be readily activated to simulate various processes, both on land and in-stream.

Scientific Detail

Land processes for pervious and impervious areas are simulated through water budget, sediment generation and transport, and water quality constituents' generation and transport. Hydrology is modeled as water balance of soil (or storage) in different layers as described by the SWM methodology. Interception, infiltration, evapotranspiration, interflow, groundwater loss, and overland flow processes are considered and are generally represented by empirical equations. Sediment production is based on detachment and/or scour from a soil matrix and transport by overland flow in pervious areas, whereas solids buildup and washoff is simulated for impervious areas. It includes agricultural components for land-based nutrient and pesticide processes and a special actions block for simulating management activities. HSPF also simulates the in-stream fate and transport of a wide variety of pollutants, such as nutrients, sediments, tracers, dissolved oxygen/biochemical oxygen demand, temperature, bacteria, and user-defined constituents.

Model Framework

- Hydrologic response unit, subwatershed, and watershed
- Simple one-dimensional stream and well-mixed reservoir/lake model

Scale

Spatial Scale

- One-dimensional, lumped parameters on a land use or subwatershed basis

Temporal Scale

- User-defined timestep, typically hourly

Assumptions

- Land simulation component is a distributed model by land use but ignores the spatial variation within a land use in a subwatershed.
- For overland flow, model assumes one-directional kinematic-wave flow.
- Model also assumes subwatershed and streams as series of reservoirs while routing flows.
- The receiving waterbody assumes complete mixing along the width and depth.

Model Strengths

- One the few watershed models capable of simulating land processes and receiving water processes simultaneously.
- Capable of simulating both peak flow and low flows.
- Simulates at a variety of timesteps, including subhourly to 1 minute, hourly or daily.

- Simulates the hydraulics of complex natural and man-made drainage networks
- Includes capabilities to address a variable water table.
- Simulates results for many locations along a reservoir or tributary.
- Includes user-defined model output options by defining the external targets block.
- Can be setup as simple or complex, depending on application, requirements, and data availability.

Model Limitations

- Relies on many empirical relationships to represent physical processes.
- Lumps simulation processes for each land use type at the subwatershed level (i.e., does not consider the spatial location of one land parcel relative to another in the watershed). The model approaches a distributed model when smaller subwatersheds are used; however, this may result in increased model complexity and simulation time.
- Requires extensive calibration.
- Requires a high level of expertise for application.
- Is limited to well-mixed rivers and reservoirs and one-directional flow.

Application History

The modeling concept had its debut in the early 1960s as the Stanford Watershed Model. During the 1970s, water quality processes were added. A FORTRAN version was developed in the late 1970s, incorporating several related models and software engineering design and development concepts funded by EPA's research laboratory in Athens, GA. In the 1980s, pre- and post-processing software, algorithm enhancements, and use of the USGS binary Watershed Data Management (WDM) system were developed jointly by the USGS and EPA. Since 1980, all model code changes have been maintained by Aqua Terra Consultants, under contract with EPA and USGS. During the mid to late 1990s, Tetra Tech, Inc., under contract with EPA developed the BASINS system and NPSM, resulting in the first Windows-based interface for the HSPF model. The current supported model release is Version 12, distributed with BASINS 3.0 as the WinHSPF model and interface. HSPF is a proven and tested continuous simulation watershed model. It is one of the models recommended by the EPA for complex TMDL studies. The HSPF model has been widely used and its application has been documented throughout its development cycle.

Model Evaluation

HSPF has been widely reviewed and applied throughout its long recent history (Hicks, 1985; Ross et al., 1997; and Tsihrintzis et al., 1996). One of the largest applications of the model was to the Chesapeake Bay Watershed, as part of the EPA's Chesapeake Bay Program's management initiative (Donigian, 1990, 1992). Tsihrintzis et al. (1994, 1995) applied HSPF in a GIS shell (using ARC/INFO) to evaluate the impact of agricultural activities, specifically transport of sediments, nutrients, and pesticides, on streams and groundwater in Southern Florida. An extensive HSPF bibliography has been compiled to document model development and application and is available online at <http://hspf.com/hspfbib.html>.

Model Inputs

- Continuous meteorological time series records including (at a minimum)
 - Rainfall
 - Potential evapotranspiration

For SNOW simulation, additional required meteorological time series include

- Temperature
- Wind speed
- Solar radiation
- Dewpoint temperature

For additional simulation options, other required meteorological time series may include

- Pan evaporation
- Cloud cover
- Soils data (auxiliary dataset to guide hydrologic calibration), pollutant buildup and washoff, stream dimensions or rating curves, and point-source loading inputs
- A large number of parameters need to be specified (some default values are available)

Users' Guide

- For model documentation, underlying theory, and parameterization, the HSPF users' manual is a recommended source (Bicknell et al., 2001).
- A browseable Windows help file version of the manual is available at <http://hspf.com/pub/hspf/HSPF.chm>.

Technical Hardware/Software Requirements

Computer hardware:

- PC

Operating system:

- DOS or Windows Operating System

Programming language:

- FORTRAN

Runtime estimates:

- Seconds to minutes or hours, depending on spatial/temporal resolution and computer performance

Linkages Supported

CE-QUAL-W2

Related Systems

WinHSPF, an interface to HSPF, is a key component of Better Assessment Science Integrating point and Nonpoint Sources (BASINS) Version 3.0. BASINS 3.0 was developed for the U.S. Environmental Protection Agency's Office of Water to respond to the continued needs of various agencies to perform watershed and water quality assessments, integrating point and nonpoint sources.

LSPC, another interface to HSPF, is available through the EPA Modeling Toolbox (<http://www.epa.gov/athens/wwqtsc/index.html>).

Sensitivity/Uncertainty/Calibration

HSPFParm is a free HSPF parameter database distributed with EPA's BASINS System. The software is installed independent of the BASINS system. It provides regionalized model parameters for published applications across the United States. It serves as a good starting point for parameter selection during model setup and calibration.

The Expert System for calibration of HSPF (HSPEXP) is an interactive program that evaluates modeled versus observed time series using over 35 rules and some 80 conditions (Lumb, 1994). It uses Artificial Intelligence techniques, incorporating expert advice, based on statistics and evaluation results, to recommend which parameters should be adjusted.

The Parameter Estimation software package (PEST) is a model calibration aid that can be run in conjunction with HSPF (Doherty, 2003). The objective function's goal is to minimize the least squares of the difference between modeled and observed flow by varying model parameters over a range that the user defines. PEST then iterates through a series of HSPF model runs, changing selected parameters and rerunning the model, until the objective is satisfied.

Model Interface Capabilities

Using the HSPF Model requires at least two files: the User Control Input File (UCI) for parameters and control specifications and a WDM file for time series. When run by itself, the UCI text file serves as the interface for the HSPF model.

The WinHSPF interface, first distributed with BASINS 3.0, provides an interactive interface to HSPF in a Windows environment. WinHSPF may be used for creating a new HSPF input sequence or for modifying an existing HSPF input sequence. The program HSPF may be run from within WinHSPF. Input sequences may be modified and saved under another name, thus creating simulation scenarios.

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C.9 HYCOM Model Description

The Hybrid Coordinate Ocean Model (HYCOM) is a hybrid-grid ocean circulation model. The South Florida Regional Model was created as an adaptation of the HYCOM model, and is hereafter called the SoFLA-HYCOM.

HYCOM Overview

Ideally, an ocean general circulation model (OGCM) should (a) retain its water mass characteristics for centuries (a characteristic of isopycnic coordinates), (b) have high vertical resolution in the surface mixed layer (a characteristic of z-level coordinates) for proper representation of thermodynamic and biochemical processes, (c) maintain sufficient vertical resolution in unstratified or weakly-stratified regions of the ocean, and (d) have high vertical resolution in coastal regions (a characteristic of terrain-following coordinates).

The hybrid coordinate is one that is isopycnal in the open, stratified ocean, but smoothly reverts to a terrain-following coordinate in shallow coastal regions, and to z-level coordinates in the mixed layer and/or unstratified seas. The hybrid coordinate extends the geographic range of applicability of traditional isopycnic coordinate circulation models (the basis of the present hybrid code), such as the Miami Isopycnic Coordinate Ocean Model (MICOM) and the Navy Layered Ocean Model (NLOM), toward shallow coastal seas and unstratified parts of the ocean. The theoretical foundation for implementing such a coordinate was set forth in Bleck and Boudra (1981) and Bleck and Benjamin (1993). In HYCOM, each coordinate surface is assigned a reference isopycnal. The model continually checks whether or not grid points lie on their reference isopycnals and, if not, tries to move them vertically toward the latter. However, the grid points are not allowed to migrate when this would lead to excessive crowding of coordinate surfaces. Thus, in shallow water, vertical grid points are geometrically constrained to remain at a fixed depth while being allowed to join and follow their reference isopycnals over the adjacent deep ocean.

In the mixed layer, grid points are placed vertically so that a smooth transition of each layer interface from an isopycnic to a constant-depth surface occurs where the interface outcrops into the mixed layer. HYCOM therefore behaves like a conventional sigma model in very shallow and/or unstratified oceanic regions, like a z-level coordinate model in the mixed layer or other unstratified regions, and like an isopycnic-coordinate model in stratified regions. In doing so, the model combines the advantages of the different types of coordinates in optimally simulating coastal and open-ocean circulation features. The present procedure of driving high-resolution coastal models (which invariably use fixed vertical grids) with output from a basin-scale isopycnic model can be streamlined, since HYCOM will be able to provide the required near-shore data at fixed depth intervals.

SoFLA-HYCOM

The SoFLA-HYCOM domain has been nested within the NA-HYCOM (North Atlantic HYCOM) that has a resolution of 1/12 degree of latitude (~6-7 km grid size), minimum depth of 20m and includes the entire North Atlantic, the Caribbean and Gulf of Mexico, as well as global thermohaline circulation (through nesting within the global HYCOM). Two types of nesting have been implemented. First a double nest, where the SoFLA-HYCOM is nested within the

IAS-HYCOM (Intra-Americas Sea) which has a 1/25 degree resolution and minimum depth of 10m. This setting was used for the climatological simulation and for sensitivity tests to assure the validity of the nesting procedure, since both IAS and SoFLA have the same resolution. The IAS nested runs also allowed the evaluation of improved flows simulated with SoFLA, due to the employment of detailed shallow topography and smaller minimum depth.

The second nesting, and final model set-up that is now used for the ongoing inter-annual simulation, is a 1/25 degree SoFLA-HYCOM nested directly to the 1/12 degree NA-HYCOM. As mentioned above, the SoFLA-HYCOM resolves the Atlantic shelf along the Florida Keys reef tract and the major passages between the Florida Keys and Florida Bay. Furthermore, the SoFLA-HYCOM includes additional sources of freshwater inputs, namely the rivers on the Southwest Florida Shelf from the Shark River to the Caloosahatchee River, which are important sources of low salinity waters that can impact the water properties around and within Florida Bay, as well as the Florida Keys through the neighboring Keys passages.

The nesting of the SoFLA-HYCOM within a larger scale HYCOM model allows the accurate simulation of the interaction between shallow water dynamics around the Florida Bay and the Florida Keys reef tract with larger scale oceanic flows. The SoFLA-HYCOM performs simulations that are suitable to provide boundary conditions to coastal hydrodynamic, water quality and ecosystem models used in Florida Bay and the Florida Keys.

The SoFLA-HYCOM domain extends from approximately 22.6°N to 27.4°N (west Florida coast) and to 26.7°N (east Florida coast) and from 78.8°W to 83.8°W . The model domain includes the south Florida coastal system and also covers the Straits of Florida between the Keys and Cuba, including the Cay Sal Bank and Bimini Island (Bahamas). The horizontal resolution is 1/25 degree (about 3 to 3.5 km in latitude) and 19 vertical layers have been implemented with isopycnal layers in the deep Straits of Florida and a combination of sigma and z-layers on the shelf areas. The bathymetry has been adopted from the 2-minute NAVO/NRL DBDB2 global dataset. The depth values in shallow areas around the Florida Keys reef tract have been corrected so that topographic details are included; this is absent in the larger scale models that provide the boundary conditions. Simulations with a 5m and a 3m minimum depth have been carried out.

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C.10 LSPC: Loading Simulation Program in C++ Model Description

Contact Information

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Download Information

Availability: Nonproprietary
<http://www.epa.gov/athens/wwqtsc/html/lspc.html>
Cost: N/A

Model Overview/Abstract

The Loading Simulation Program in C++ (LSPC) is a watershed modeling system that includes streamlined Hydrologic Simulation Program FORTRAN (HSPF) algorithms for simulating hydrology, sediment, and general water quality on land as well as a simplified stream transport model. It is a EPA-accepted TMDL modeling application, developed by Tetra Tech, Inc., partially under contract with EPA. A key advantage of LSPC is that it has no inherent limitations in terms of modeling size or model operations and has been applied to large, complex watersheds. In addition, the Microsoft Visual C++ programming architecture allows for seamless integration with modern-day, widely available software such as Microsoft Access and Excel.

Model Features

- Watershed modeling
- Hydrologic simulation and hydraulic transport
- Overland and in-stream sediment simulation
- Temperature simulation

Model Areas Supported

Watershed	High
Receiving Water	Medium
Ecological	None
Air	Medium
Groundwater	Medium

Model Capabilities

Conceptual Basis

In LSPC, a subwatershed is typically conceptualized as a group of various land uses all routed to a representative stream segment. Several small subwatersheds and representative streams may be networked together to represent a larger watershed drainage area. Various modules are available and may be readily activated to simulate various land and stream processes.

Scientific Detail

Land processes for pervious and impervious areas are simulated through water budget, sediment generation and transport, and water quality constituents' generation and transport. Hydrology is modeled as water balance of soil (or storage) in different layers as described by the Stanford Watershed Model (SWM) methodology. Interception, infiltration, evapotranspiration, interflow, groundwater loss, and overland flow processes are considered and are generally represented by empirical equations. Sediment production is based on detachment and/or scour from a soil matrix and transport by overland flow in pervious areas, whereas solids buildup and washoff is simulated for impervious areas. For water quality, buildup and washoff of a quality constituent on a land surface, baseflow-associated pollutant concentrations, sediment-associated pollutant load, and soil temperature and heat transfer to water are available.

For in-stream simulation, the model simulates radiation and heat transfer, conservative substance routing, suspended solids routing and settling, and general first-order pollutant loss, which is applicable for simulating a wide range of conservative substances, such as fecal coliform bacteria, total nitrogen and phosphorus, and total metals. The hydraulic compartment, which determines the advection terms for all of the other components, is based on a time-interval budget of water volume between inflow from the above reach, user-specified outflows, and discharge to the next downstream reach.

Model Framework

- Watershed hydrologic, sediment, and water quality
- Time series climate-driven, overland, subsurface, and in-stream simulation

Scale

Spatial Scale

- Lumped parameters at a land use subwatershed scale
- One-dimensional in-stream fate and transport
- Capable of simulating many subwatersheds (100+) over large drainage areas (8-digit HUCs)

Temporal Scale

- User-defined timestep, typically hourly

Assumptions

- Land simulation component is a distributed model by land use but ignores the spatial variation within a land use in a subwatershed.

- For overland flow, model assumes one-directional kinematic-wave flow.
- Model also assumes subwatershed and streams as series of reservoirs while routing flows.
- Model assumes complete mixing along the width and depth of the receiving waterbody.

Model Strengths

LSPC is one the few watershed models that is capable of simulating land processes and receiving water processes simultaneously. It is capable of simulating both peak flow and low flows and a variety of timesteps, including sub-hourly to one minute, hourly or daily. The model simulates the hydraulics of complex natural and man-made drainage networks and includes capabilities to address variable water table. A key strength of LSPC model output is that the model automatically aggregates results and manages the output at a subwatershed or reach-segment level. The model can be setup as simple or complex, depending on the application requirement and data availability. The design of the modeling system and supporting databases is particularly well suited for efficient application to large, complex watersheds. Data management tools support the evaluation of loading and management within multiple watersheds simultaneously.

Model Limitations

The model relies on many empirical relations to represent physical processes. For land simulation, processes are lumped for each land use type at the subwatershed level; therefore, the model does not consider the spatial location of one land parcel relative to another in the watershed. The model approaches a distributed model when smaller subwatersheds are used; however, this may result in increased model size and simulation time. For in-stream simulation, it is limited to well mixed rivers and reservoirs and one-directional flow. It requires extensive calibration. It generally requires a high level of expertise for application.

Application History

LSPC is the key watershed modeling component of the TMDL Toolbox. TMDL's model applications have been successfully developed using LSPC in Alabama, Mississippi, South Carolina, Georgia, California, Kentucky, Tennessee, West Virginia, Virginia, Maryland, Arizona, Ohio, Montana, Puerto Rico, and U.S. Virgin Islands. Several recent watershed TMDLs have been done using LSPC. Two are listed as follows:

- Published Tennessee TMDL documents may be accessed at the following web address:
<http://www.state.tn.us/environment/wpc/tmdl/approved.php>
- Published Alabama TMDL documents may be accessed at the following web address:
<http://www.epa.gov/region4/water/tmdl/alabama/index.htm>

Model Evaluation

HSPF, on which LSPC is based, has been widely reviewed and applied throughout its history (Hicks, 1985; Ross, et al., 1997; and Tsihrintzis, et al., 1996). Tsihrintzis, et al., (1994, 1995) applied HSPF in a GIS shell (using ARC/INFO) to evaluate the impact of agricultural activities, specifically transport of sediments, nutrients, and pesticides, on streams and groundwater in Southern Florida. The underlying HSPF algorithms used in the LSPC model have been widely used and the applications have been documented over more than 20 years.

Model Inputs

- Continuous meteorological time series records including (at a minimum)
 - Rainfall
 - Potential evapotranspiration

For SNOW simulation, additional required meteorological time series include

- Temperature
- Wind speed
- Solar radiation
- Dewpoint temperature
- Soils data (auxiliary dataset to guide hydrologic calibration), pollutant buildup and washoff, stream dimensions or rating curves, and point-source loading inputs
- A large number of parameters need to be specified (some default values are available)

Users' Guide

- An LSPC Users' Manual (Tetra Tech, 2002) is available through EPA's Watershed and Water Quality Modeling Technical Support Center.
- For model documentation, underlying theory, and parameterization, the HSPF users' manual is a recommended source (Bicknell, et al., 2001).
- A browseable Windows help file version of the manual is available at <http://hspf.com/pub/hspf/HSPF.chm>.

Technical Hardware/Software Requirements

Computer hardware:

- PC (Pentium III or higher recommended, but not required)

Operating system:

- Windows 98 or later

Programming language:

- C++

Runtime estimates:

- Seconds to minutes or hours, depending on spatial/temporal resolution

Linkages Supported

EFDC, WASP, CE-QUAL-W2, EPD-Riv1

Related Systems

HSPF, WinHSPF

Sensitivity/Uncertainty/Calibration

LSPC includes a data analysis component that may be used to quickly compare model output against observed data in time series form, as monthly summaries, or on a one-to-one graph.

LSPC model output is especially tailored for spreadsheet use; consequently, many users prefer to

develop independent spreadsheet analysis, summarization, calibration, and plotting applications, which are readily linked to LSPC model output.

HSPFParm is a free HSPF parameter database distributed with EPA's BASINS System. The software is installed independent of the BASINS system. It provides regionalized model parameters for published applications across the United States. It serves as a good starting point for parameter selection during model setup and calibration.

The Parameter Estimation software package (PEST) is a model calibration aid that can be run in conjunction with HSPF (Doherty, 2003). Although PEST is not currently linked with LSPC, it is a generalized calibration system that can potentially be linked to any model. Since parameter values and definitions in HSPF and LSPC are interchangeable, results from a small-scaled HSPF PEST run can be used to calibrate LSPC. The objective function's goal is to minimize the least squares of the difference between modeled and observed flow by varying model parameters over a range that the user defines. PEST then iterates through a series of model runs until the objective is satisfied.

Model Interface Capabilities

When launched from within the EPA Watershed Characterization System (WCS), LSPC has an extension for automatically setting up the model using GIS information. The LSPC model interface has two components: (1) a stand alone GIS component, and (2) a model parameter management component. The LSPC Map Objects GIS interface, which is compatible with ArcView shapefiles, acts as the control center for managing and launching watershed model scenarios. The stand alone interface communicates with shapefiles and the underlying Access database. Therefore, once a watershed system is created, it is easily transferable to users who may not have ArcView or MS Access. The LSPC model parameter management component can be used either independently on previously saved model setup files or in conjunction with the GIS component to setup model scenarios on the fly.

LSPC also includes a suite of additional tools, including data management tools for editing watershed data, data inventory tools for reporting and summarizing model inputs and outputs, and data analysis tools for both visualizing and summarizing model inputs or outputs and observed data. The model includes a TMDL calculation that allows the user to specify source-specific allocations and generate corresponding model results for TMDL analysis.

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C.11 PLSM: Pollutant Load Screening Model Description

The Pollutant Load Screening Model (PLSM) is a watershed model created by St Johns River Water Management District (SJRWMD) staff to support the 1995 District Water Management Plan. PLSM (Adamus and Bergman 1995) utilizes a GIS framework to calculate constituent loads as the product of water quality concentrations associated with certain land use practices, and runoff water volume associated with those same practices.

The PLSM was originally developed using ARC Macro Language (AML) in ArcInfo 7.2 GRID. Since PLSM was developed in the 1990s, ArcInfo has undergone significant interface and data management changes. AML has been replaced as the main programming language for tool development with other languages such as Visual Basic or Python. The district has fully implemented the changes in ArcInfo and new ArcGIS tools aid the use of PLSM modeling. Of particular interest is ModelBuilder, a graphic modeling framework for designing and implementing geoprocessing models that can include tools, scripts, and data. It was incorporated in ArcGIS at version 9.0 and provides a good transition program for replicating the processes of PLSM.

Models are easily created in ArcToolbox, and existing models can be customized by any user. ModelBuilder works with both vector and raster data files. Users can create documentation for each process and have easy access to help files.

The PLSM uses five input grids: existing land use, drainage basin boundaries, rainfall, soils, and future land use as well as several remap tables (i.e., tables used to reclassify grid data). All the input files must be prepared prior to running the model, which executes without user interaction.

The overall process starts with the calculation of runoff for each land use and soils combination within the study area. Runoff is determined by multiplying average annual rainfall, a runoff coefficient that depends on soil and land use type, and the area of the basin under study. In the next step, the annual pollutant load is calculated for different parameters such as total phosphorus or total nitrogen. Annual load is determined by multiplying runoff by a runoff pollutant concentration coefficient that depends on the type of land use. The last step adds storm water treatment efficiencies to runoff. Storm water treatment facilities are designed to improve the quality of runoff before it enters water bodies, and their impact is added to PLSM through efficiency values.

The AML outputs annual load grids that let users visualize the areas that contribute different pollutants and a summary of the annual loads by drainage basin. When PLSM was developed in the 1990s it was run for every drainage basin in the district.

References

Adamus, C.L. and M.J. Bergman. 1995. Estimating nonpoint source pollution loads with a GIS screening model. *Wat. Res. Bull.* 31(4):647–655.

C.12 POM: Princeton Ocean Model Description

The Princeton Ocean Model (POM) was developed by Mellor and Blumberg for application to oceanographic problems in the Atmospheric and Oceanic Sciences Program of Princeton University, the Geophysical Fluid Dynamics Laboratory of NOAA and Dynalysis of Princeton.

The principal attributes of the model are as follows:

- It contains an imbedded second moment turbulence closure sub-model to provide vertical mixing coefficients.
- It is a sigma coordinate model in that the vertical coordinate is scaled on the water column depth.
- The horizontal grid uses curvilinear orthogonal coordinates and an "Arakawa C" differencing scheme.
- The horizontal time differencing is explicit whereas the vertical differencing is implicit. The latter eliminates time constraints for the vertical coordinate and permits the use of fine vertical resolution in the surface and bottom boundary layers.
- The model has a free surface and a split time step. The external mode portion of the model is two-dimensional and uses a short time step based on the CFL condition and the external wave speed. The internal mode is three-dimensional and uses a long time step based on the CFL condition and the internal wave speed.
- Complete thermodynamics have been implemented.

The sigma coordinate system is probably a necessary attribute in dealing with significant topographical variability such as that encountered in estuaries or over continental shelf breaks and slopes. Together with the turbulence sub-model, the model produces realistic bottom boundary layers which are important in coastal waters (Mellor 1985) and in tidally driven estuaries (Oey et al. 1985a, b), which the model can simulate since it does have a free surface. The bottom boundary layers are also important for deep water formation processes (Zavatarelli and Mellor 1995; Baringer and Price 1997; Jungclaus and Mellor 2000; Ezer and Mellor 2004) and for the maintenance of the baroclinicity of oceans basins (Mellor and Wang 1996).

The specific program that is now supplied to outside users simulates the flow, east to west across a seamount with a prescribed vertical temperature stratification, constant salinity, zero surface heat and salinity flux and a zero wind stress distribution although wind stress may be easily applied. The program should run with no additional data requirements. The open boundary conditions specified for this problem are a sampling of many possible open boundary conditions. Users may define their own problems, defined by topography, horizontal grid, vertical sigma grid and boundary conditions. The model can be downloaded at <http://www.aos.princeton.edu/WWWPUBLIC/htdocs.pom/>.

References

Baringer, M. O., and J. F. Price. 1997. Momentum and energy balance of the Mediterranean outflow. *Journal of Physical Oceanography* 27(8):1678–1692.

- Ezer, T. and G. L. Mellor, 2004: A generalized coordinate ocean model and a comparison of the bottom boundary layer dynamics in terrain-following and in z-level grids. *Ocean Modelling*, 6, 379-403.
- Jungclaus, J.H. and Mellor, G.L., 2000: A three-dimensional model study of the Mediterranean outflow, *J. Mar. Syst.*, 24, 41–66.
- Mellor, G. L. and X. H. Wang, 1996: Pressure compensation and the bottom boundary layer, *J. Phys. Oceanogr.*, 26(10), 2214-2222, 1996.
- Mellor 1985
- Oey et al. 1985a, b (there is only one Oey et al., 1985 listed in the references
- Oey, L-Y., G. L Mellor, and R. I. Hires, 1985: Tidal modeling of the Hudson-Raritan Estuary. *Estuarine, Coastal and Shelf Science*, 20, 511-527.
- Zavatarelli M., Mellor G. L. (1995) A numerical study of the Mediterranean Sea circulation. *Journal of Physical Oceanography*, 25(6), 1384-1414.

C.13 SPARROW: SPAtially Referenced Regression On Watershed Attributes Model Description

Contact Information

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Other Contacts:

NATIONAL SPARROW (NAWQA Nutrient Synthesis, Reston, VA)

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Schwarz, Gregory	703-648-5718 (gschwarz@usgs.gov)
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REGIONAL SPARROW

Chesapeake Bay (Maryland District)

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Preston, Steve	410-267-9875 (spreston@usgs.gov)

New England (New Hampshire District)

Johnston, Craig	603-226-7843 (cmjohnst@usgs.gov)
Moore, Richard	603-226-7825 (rmoore@usgs.gov)
Robinson, Keith	603-226-7809 (kwrobins@usgs.gov)

North Carolina Coastal (North Carolina District)

McMahon, Gerard	919-571-4068 (gcmcmahon@usgs.gov)
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Download Information

Availability: Nonproprietary

Cost: N/A

Model Overview/Abstract

SPARROW relates in-stream water quality measurements to spatially referenced characteristics of watersheds, including contaminant sources and factors influencing terrestrial and stream transport. The model empirically estimates the origin and fate of contaminants in streams and quantifies uncertainties in these estimates based on model coefficient error and unexplained variability in the observed data.

Model Features

- Empirically based method
- Riverine pollutant loading rates prediction

- Contaminants modeled: sediment, nutrients, etc.
- Datasets used: Reach File (RF1 or version), USGS's National Land Cover Dataset (NLCD), STATSGO, and other spatial datasets

Model Areas Supported

Watershed	Medium
Receiving Water	Low
Ecological	None
Air	Low (Model supports inclusion of atmospheric deposition loads)
Groundwater	None

Model Capabilities

Conceptual Basis

The SPARROW model uses spatially referenced regressions of contaminant transport on watershed attributes to support regional water quality assessment goals, including descriptions of spatial and temporal patterns in water quality and identification of the factors and processes that influence those conditions. The method is designed to reduce the problems of data interpretation caused by sparse sampling, network bias, and basin heterogeneity.

Scientific Detail

SPARROW uses statistical methods to calibrate a simple, structural model of riverine water quality, one that imposes mass balance in accounting for changes in contaminant flux. Regression equations relate measured transport rates in streams to spatially referenced descriptors of pollution sources and land surface and stream channel characteristics. Spatial referencing of land-based and water-based variables is accomplished via superposition of a set of contiguous land surface polygons on a digitized network of stream reaches that define surface water flow paths for the region of interest. The primary spatial reference frame for the model is the RF1 reach network: all point sources and landscape features are referenced to a particular RF1 reach.

Water quality measurements are obtained from monitoring stations located in a subset of the stream reaches. Water quality predictors in the model are developed as a function of both reach and land surface attributes and include quantities describing contaminant sources (point and nonpoint) as well as factors associated with rates of material transport through the watershed (such as soil permeability and stream velocity). Predictor formulae describe the transport of contaminant mass from specific sources to the downstream end of a specific reach. Loss of contaminant mass occurs during both overland and in-stream transport. The model can also take into account pollutant loads contributed by atmospheric deposition.

SPARROW was first used to estimate the distribution of nutrients in streams and rivers of the U.S. and subsequently shown to describe land and stream processes affecting the delivery of nutrients (Smith, et al., 1997; Alexander, et al., 2000; Preston and Brakebill 1999).

Model Framework

- Empirical, regression-based
- Uses national datasets, wide applicability

Scale

Spatial Scale

- Large watersheds

Temporal Scale

- Annual
- User-defined modeling period

Assumptions

The model is based on an empirical regression approach using mass balance calculations. Regression equations relate measured transport rates in streams to spatially referenced descriptors of pollution sources and land surface and stream channel characteristics.

Model Strengths

The model is capable of simulating a variety of pollutants at different spatial scales using national level datasets, including RF1 (stream reach file), NLCD (USGS land use/land cover), and STATSGO (NRCS soils data). The model can be used to model large- and small-scale systems with flexibility in the datasets and level of detail incorporated. The model is readily available from USGS and has been applied in several case studies.

Model Limitations

The model is limited to broadly estimating pollutant loads and fate/transport characteristics. Stream processes and model output are based on statistical relationships that were developed using national and regional water quality datasets.

Application History

The model has been primarily used to estimate nutrient and sediment loads at various spatial scales. Refer to the USGS SPARROW website for case studies.

Model Evaluation

Refer to USGS website - <http://water.usgs.gov/nawqa/sparrow/>

Model Inputs

- Initial conditions
- Time sequences of boundary conditions (inputs from watershed sources and discharges)
- Stream reach file reference (e.g., RF1)
- Physical coefficients
- Biological and chemical reaction rates
- Land use, soils, and other spatial datasets

Users' Guide

Not readily available on website. Website provides several journal articles and contacts:

<http://water.usgs.gov/nawqa/sparrow/>

Technical Hardware/Software Requirements

Computer hardware:

- PC

Operating system:

- PC-DOS

Programming language:

- SAS

Runtime estimates:

- Minutes

Linkages Supported

None

Related Systems

None

Sensitivity/Uncertainty/Calibration

In calibrating the model, measured rates of contaminant transport are regressed on predicted transport rates at the locations of the monitoring stations, giving rise to a set of estimated linear and nonlinear coefficients from the predictor formulae.

Once calibrated, the model is used to estimate contaminant transport and concentration in all stream reaches. A variety of regional characterizations of water quality conditions are then possible based on statistical summarization of reach-level estimates. The application of bootstrap techniques allows estimation of the uncertainty of model coefficients and predictions.

Model Interface Capabilities

None

References

Refer to USGS website for complete list - <http://water.usgs.gov/nawqa/sparrow/>

Smith, R.A., G.E. Schwarz, and R.B. Alexander. 1997. *Regional Interpretation of Water-quality Monitoring Data*. Water Resources Research, 33(12): 2781-2798.

Alexander, R.B., R.A. Smith, M.J. Focazio, and M.A. Horn. 1999. *Source-Area Characteristics of Large Public Surface-Water Supplies in the Conterminous United States: An*

Information Resource for Source-Water Assessment. Open-File Report 99-248. U.S. Geological Survey, Reston, VA.

Preston, S.D. and J.W. Brakebill. 1999. *Application of Spatially Referenced Regression Modeling for the Evaluation of Total Nitrogen Loading in the Chesapeake Bay Watershed*. Report 99-4054. U.S. Geological Survey Water Resources Investigations, Baltimore, MD.

C.14 WAMView: Watershed Assessment Model with an ArcView Interface Model Description

Contact Information

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<http://www.swet.com>

Download Information

Availability: Proprietary.
Cost: N/A

The source code is not available. The executable code can be downloaded from
<http://www.swet.com> with a registration.

Model Overview/Abstract

WAMView allows engineers and planners to assess the water quality of surface and groundwater based on land use, soils and weather stations.

Model Features

- Overland attenuation
- Stream routing and various water control structures

Model Areas Supported

Watershed	High
Receiving Water	None
Ecological	Low
Air	None
Groundwater	Low

Model Capabilities

Conceptual Basis

In WAMView, the watershed is conceptualized as a series of cells/grids with different land use, soil, and land slope. The streams are conceptualized as a series of reaches with computed geometries based on upstream drainage areas, which may be redefined by the user.

Scientific Detail

The watershed runoff model BUCSHELL generates grid-based runoff based on land use, soil, topography, and rainfall with two built-in watershed model GLEAMS and EAAMOD. GLEAMS is applied to upland while EAAMOD is applied to the area with a shallow groundwater table.

The stream routing algorithm BLASROUTE is developed based on Manning's equation without a momentum component. A looping technique is implemented in the stream routing algorithm, and various flow structure configurations and operation schedules are implemented. It is noted that BLASROUTE employs first order attenuation from cells to reaches, depression and wetlands.

Model Framework

- Grid-based watershed
- One-dimensional stream routing

Scale

Spatial Scale

- Grid-based watershed; typical grid size 100m x 100m
- Typical reach/stream length 1000m to 10000m

Temporal Scale

- User-defined timestep: typically, a day

Assumptions

- Transport of water and constituent is dependent on flow distances, gradients, and type of conveyance system.
- All input data such as land use, soil, hydrology coverages, and land use management activities are accurate.
- Rainfall data from individual stations are representative of rainfall across the entire basin.
- A reservoir routing technique without a momentum component is representative of low gradient streams.
- Phosphorous and nitrogen process models within the submodels are representative of actual transport processes.

Model Strengths

- Capable of simulating the water quality of surface and groundwater based on land use, soil and weather.
- Capable of simulating various BMP scenarios.
- Provides a higher resolution of results than models that rely on polygon coverages.
- Works well for wetlands.
- Capable of routing attenuated runoff into a complex reach network with flow structures in the latest version.

Model Limitations

- Does not include a momentum component in the stream routing algorithm.
- May predict flow inaccurately when applied to streams with steep slopes.
- Considers limited chemistry constituents.
- Includes groundwater component empirically, not fully integrated into the system.

Application History

Past applications of WAMView include:

- St. Johns River Watershed Assessment Project
- Suwannee River Watershed Assessment Project
- Lower St. Johns River Mainstem Subbasins Hydrologic/Water Quality Modeling
- Hydrologic Water Quality Assessment for Myakka River Basin
- North Palm Beach County Basin Pollutant Loading and Abatement Analysis

Model Evaluation

Model evaluation was done through various projects in which the model was calibrated and then validated to a different time period. The two watershed models used in WAMView are GLEAMS (modified slightly by SWET) and EAAMOD (developed by SWET for areas with a shallow groundwater table). They were tested and evaluated in the past. (See EAAMOD -- Everglades Agricultural Area Model at <http://www.swet.com/>.)

Model Inputs

- Time sequences of boundary conditions—outflow with chemistry constituents of interest (if tidal outflow exits), point sources flow with chemistry constituent of interest
- Basin polygon coverage, topography coverage, land use coverage, soil coverage, reach coverage, rain station coverage, and utility coverage if any
- Time sequences of rainfall data for each station and other weather data, including monthly maximum and minimum temperature, monthly average dew point temperature, wind speed and solar radiation
- Water control structure configurations and operation schedules, if any

Users' Guide

Available with the download of the executable code: <http://www.swet.com>. The guide is too simple.

Technical Hardware/Software Requirements

Computer hardware:

- Minimum 100 Mb hard disk space
- Minimum 64 Mb RAM
- Minimum 200 MHZ co-processor
- Minimum 600 x 800 screen resolution

Operating system:

- Windows 95/98/ME/NT/2000/XP
- ArcView 3.2a with Spatial Analyst 1.1 (or 2.0)

Programming language:

- FORTRAN for BUCSHELL and BLASROUTE
- AVENUE for pre- and post-processor in a customized ArcView

Runtime estimates:

- Hours to a day

Linkages Supported

None

Related Systems

None

Sensitivity/Uncertainty/Calibration

Not available.

Model Interface Capabilities

The model interface is a customized ArcView interface with good pre- and post-processors. The pre-processor can process GIS coverage data and create some model input files, and the post-processor can display simulated results in GIS view and layout windows.

References

WAMView User's Manual. Soil and Water Engineering Technology, Inc.

Jacobson, B.M., A.B. Bottcher, N.B. Pickering, and J. G. Hiscock. 1998. Unique routing algorithm for watershed assessment model. *American Society of Agricultural Engineers Paper* No. 98-2237. American Society of Agricultural Engineers, St. Joseph, MI.

Bottcher, A.B., J.G. Hiscock, N.B. Pickering, and B.M. Jacobson. 1998. WAM: Watershed Assessment Model for Agricultural and Urban Landscapes. Presented at the 7th International Conference on Computers in Agriculture. October 26-30, 1998, Orlando, FL.

C.15 WASP: Water Quality Analysis Simulation Program Model Description

Contact Information

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Download Information

Availability: Nonproprietary
Version 5.1: <http://www.epa.gov/ceampubl/swater/wasp/index.htm>
Version 6.2: <http://www.epa.gov/athens/wwqtsc/html/wasp.html>
Cost: N/A

Model Overview/Abstract

WASP is a generalized modeling framework based on the finite-volume concept for quantifying fate and transport of water quality variables in surface waters. The three components of the model are WASP for mass transport; EUTRO for dissolved oxygen, nutrients, and algal kinetic; and TOXI for toxic substances. WASP is capable of analyzing time-variable or steady state, one-, two-, or three-dimensional water quality problems. WASP5 is a DOS application and WASP6 is a Windows application. WASP model has been widely applied to investigate dissolved oxygen, bacteria, eutrophication, suspended solids, and toxic substance problems.

Model Features

- Tracer transport
- Eutrophication
- Dissolved oxygen
- Nutrients
- Toxic
- Sediment transport

Model Areas Supported

Watershed	None
Receiving Water	High
Ecological	Medium
Air	None
Groundwater	None

Model Capabilities

Conceptual Basis

The waterbodies are conceptualized as well-mixed control volume, and the law of conservation of mass is applied to each control volume.

Scientific Detail

The governing equations for the WASP model are the advection-dispersion-reaction equations for the water quality variables. The WASP module provides the advection and dispersion solution and the EUTRO and TOXI modules provide the reaction solutions of dissolved oxygen, nutrients, and algae. A one-step Euler solution technique is applied for the time difference. The advection terms in the governing equations are solved with UPWIND difference in space. The water quality variables can be turned on/off depending on modeling requirements.

Model Framework

- One-, two-, or three-dimensional
- Any type of waterbody

Scale

Spatial Scale

- One-, two-, or three-dimensional

Temporal Scale

- User-defined timestep

Assumptions

- Completely mixing control volume

Model Strengths

- WASP model is a very flexible modeling framework and can simulate water quality in one-, two-, or three-dimensional space.
- The control volume structure promises the conservation of mass. WASP provides the transport computation framework and can be incorporated with EUTRO to simulate eutrophication, nutrient, and dissolved oxygen. It also can be incorporated with TOXI to model metals, toxics, and sediment transport.

Model Limitations

- Requires external hydrodynamic model to provide flow file for solving advection. The file size might be very large in several gigabytes for long-term simulation.
- User specified dispersion coefficient and temperature.
- First-order UPWIND difference in space may cause significant numerical diffusion.
- Over-simplified sediment flux calculation.
- No periphyton or macroalgae.
- Sediment transport processes are not related to shear stress.

Application History

A significant amount of WASP applications can be found in technical reports, journal and conference papers. Examples of application include modeling eutrophication of Tampa Bay, Neuse River, the Great Lakes, and Potomac Estuary; and examining phosphorus loading to Lake Okeechobee, PCB pollution of the Great Lakes, and kepone pollution of the James River Estuary.

Model Evaluation

WASP model is widely cited in peer reviewed journal papers.

Model Inputs

- Initial conditions
- Point and nonpoint sources inputs
- Flow file
- Vertical mixing coefficients
- Open boundary conditions
- Biological and chemical reaction rates

Users' Guide

WASP5:

- The Water Quality Analysis Simulation Program, WASP5, Part A: Model Documentation.
- The Water Quality Analysis Simulation Program, WASP5, Part B: The WASP5 Input Data Set
- Available in model download file: <http://www.epa.gov/ceampubl/swater/wasp/index.htm>

WASP6:

- Water Quality Analysis Simulation Program (WASP) Version 6.0, Draft: User's Manual
- Available online: <http://www.epa.gov/athens/wwqtsc/html/wasp.html>:

Technical Hardware/Software Requirements

Computer hardware:

- PC

Operating system:

- PC-DOS, Windows

Programming language:

- FORTRAN (WASP5),

Runtime estimates:

- Minutes to hours

Linkages Supported

DYNHYD5 provides the flow information to WASP5. Other models that provides flow file include RIVMOD, EFDC, and SWMM.

Related Systems

QUAL2E, QUAL2K, CE-QUAL-RIV1, CE-QUAL-W2, CW-QUAL-ICM, CAEDYM

Sensitivity/Uncertainty/Calibration

Not available

Model Interface Capabilities

- WASP6 provides a Windows interface including pre-processor and post-processor

References

- Ambrose, R.B., T.A. Wool, and J.L. Martin. 1993a. *The Water Quality Analysis Simulation Program, WASP5, Part A: Model Documentation*. U.S. Environmental Protection Agency, Center for Exposure Assessment Modeling, Athens, GA.
- Ambrose, R.B., T.A. Wool, and J.L. Martin. 1993b. *The Water Quality Analysis Simulation Program, WASP5, Part B: The WASP5 Input Dataset*. U.S. Environmental Protection Agency, Center for Exposure Assessment Modeling, Athens, GA.
- Wool, T.A., R.B. Ambrose, J.L. Martin, and E.A. Comer. *Water Quality Analysis Simulation Program (WASP) Version 6.0 Draft: User's Manual*

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Appendix D. Division of Estuarine and Coastal Waters into Subbasins for Nutrient Criteria Formulation

The watershed-based approach for identifying waterbodies described in Chapter 3 for estuaries is less suited to conditions in South Florida due to South Florida's highly managed inland flows and open-water dominated systems (i.e., Florida Bay and Keys). Yet South Florida's marine waters have been the subject of long-term (>15 years) monitoring as part of the Southeast Environmental Research Center (SERC) water quality monitoring network. In cooperation with EPA, scientists for the National Park Service and Florida International University used principal component and cluster analysis to segment South Florida marine waters based on their similarities and differences. The SERC data are unique to South Florida and are appropriate for principal component and cluster analysis because they provide a long-term record with consistent sampling and analytical methods. This appendix contains a report from these scientists and their proposed segmentation for South Florida's marine waters. EPA provides a further analysis of the background information provided by these scientists in Section 5.5 and is considering using the proposed segmentation for developing nutrient criteria in South Florida's marine waters.

MEMORANDUM

To: Jim Hagy (EPA), Ken Weaver (FDEP), Eva DiDonato (NPS)

Cc: Henry Briceño (FIU), Susan Markley (DERM), Denesia Cheek (NPS)
Sarah Bellmund (BNP)

From: Joffre Castro (ENP)

Re: Division of Estuarine and Coastal Waters into
Subbasins for Nutrient Criteria Formulation

Date: May 10, 2010

Everglades National Park (ENP) and Florida International University (FIU) propose that estuarine and coastal waters in south Florida be divided into subbasins, with unique water characteristics, for the purpose of deriving nutrient criteria. On behalf of ENP, FIU has completed an extensive statistical characterization of coastal waters from Biscayne Bay to Dry Tortugas to Pine Island Sound. The analyses, a combination of Principal Component Analysis and Hierarchical Clustering of multiple (from 8 to 13) parameters, produced a division of the bays as shown in the figure and table below. Attached is an abbreviated summary of the analysis prepared by Henry Briceño, Joe Boyer, and Peter Harlem from FIU.

We are proposing that the Environmental Protection Agency (EPA) and Florida Department of Environmental Protection (FDEP) adopt this division of the bays for establishing nutrient criteria in estuarine and coastal waters in south Florida. Although the ENP / FIU analysis for proposing numeric nutrient criteria is only partially completed, we recognize the importance of sharing this information with EPA and FDEP as soon as possible. During the next several weeks, we'll continue examining these subbasins and estimating summary nutrient statistics. These statistics will be the basis for proposing numeric criteria for each subbasin. A final step will include testing the significance of these criteria with the purpose of combining subbasins with similar criteria (not statistically different) for total phosphorus, total nitrogen, and chlorophyll-a.

The GIS layers with information on the subbasins as well as FIU's working files with data and statistical analysis are available at a FIU's ftp site. We'll provide to EPA, FDEP, and other stakeholders accessing and log-in information via email. After EPA and FDEP review the data, we will convene a meeting to discuss the proposed subbasins.

On the basis of the statistical analysis, the bays and coastal waters were divided as follows:

Biscayne Bay was divided into nine subbasins: two north of the Rickenbacker Causeway, six south of the Rickenbacker Causeway, and one that includes both Barnes Sound and Manatee Bay.

Florida Bay was divided into six subbasins: one on the eastern side of the bay, one along the northern shore that is a transitional zone made up of several small bays, one that includes the many inshore coastal lakes, two on the center of the bay (central and southern), and one on the western side of the

bay. It is important to point out that the Coastal Lakes subbasin is currently considered provisional. We're in the processes of collecting pertinent water quality data to validate and confirm this grouping.

The Florida Keys region was divided into five subbasins: one on the ocean side along the reef track, two on the Gulf side, one for the Marquesas, and one for Dry Tortugas.

The Gulf Shelf was divided into three subbasins: inner, middle, and outer. These subbasins include the area west of Florida Bay, the coastal waters around Cable Sable, and the waters west of the Ten Thousand Islands.

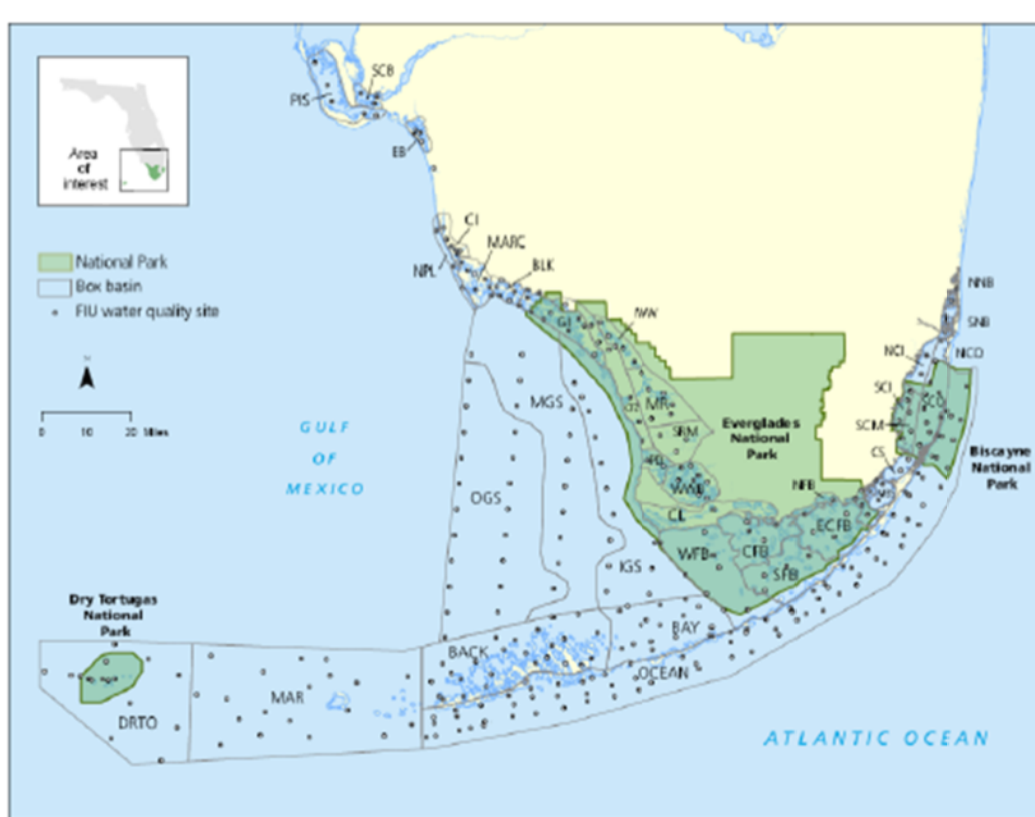


Figure. Proposed subdivision of estuarine and coastal waters for deriving nutrient criteria.

Methods and Approaches for Deriving Numeric Criteria for Nitrogen/Phosphorus Pollution in Florida's Estuaries, Coastal Waters, and Southern Inland Flowing Waters

Table. Subbasin information by region

Region	No.	Key	Subbasin
BISCAYNE BAY	1	NNI	Northern North Bay
	2	SNB	Southern North Bay
	3	NCO	North Central Outer Bay
	4	NCI	North Central Inshore
	5	SCO	South Central Outer Bay
	6	SCI	South Central Inshore
	7	SCM	South Central Mid-Bay
	8	CS	Card Sound
	9	MIB	Macatee Bay and Dames Sound
FLORIDA BAY	1	BCFI	East-Central Florida Bay
	2	NFB	Northern Florida Bay
	3	CL	Coastal Lakes
	4	CFB	Central Florida Bay
	5	SFB	Southern Florida Bay
	6	WFB	Western Florida Bay
FLORIDA KEYS	1	OCSAN	Oceanside
	2	BAY	Bayside
	3	BACK	Backcountry
	4	MAR	Marquesas
	5	DRTO	Dry Tortugas

Region	No.	Key	Subbasin
GULF SHELF	1	IBS	Inner Gulf Shelf
	2	MCB	Middle Gulf Shelf
	3	OCS	Outer Gulf Shelf
WHITEWATER BAY TO TEN THOUSAND ISS.	1	WWB	Whitewater Bay
	2	PDL	Ponce de Leon
	3	SRM	Shark River mouth
	4	MRZ	Mangrove River Zone
	5	CTZ	Coastal Transition Zone
	6	IWW	Inner Waterway
	7	GI	Gulf Islands
	8	BLR	Blackwater River
PINE IS. TO ROOKERY BAY	1	MAR	Marco Island
	2	NPL	Naples Bay
	3	CI	Collier Inshore
	4	EB	Estero Bay
	5	SCB	San Carlos Bay
	6	PIS	Pine Island Sound

The Whitewater Bay to Ten Thousand Islands region was subdivided into eight subbasins: Whitewater Bay, Ponce de Leon Bay, Shark River Mouth, Mangrove River Zone, Coastal Transition Zone, Inner Waterway, Gulf Islands, and Blackwater River.

The Pine Island to Rookery Bay region was divided into six subbasins: Marco Island, Naples Bay, Collier Inshore, Estero Bay, San Carlos Bay, and Pine Island Sound. This region is completely outside park boundaries but is included to provide a general picture of bays around and adjacent to national park lands.

**Summary of Statistical Classification and Clustering of South Florida
Estuarine and Coastal Waters**

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Miami April 30th, 2010

TASK AGREEMENT #: J5297-08-0085 COOPERATIVE AGREEMENT#: H5000-06-0104

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PREVIOUS WORK:

Classification and grouping of South Florida coastal waters into spatial WQ clusters have been performed by Boyer et al. (1997) and Briceño and Boyer (2010) in Florida Bay (FB); by Caccia and Boyer (2005), Hunt and Todt (2006), and Briceño et al. (2010) in Biscayne Bay (BB); by Boyer (2006) in the Whitewater Bay-Ten Thousand Islands (WWB-TTI) region; and by Boyer and Briceño (2009) in the Florida Keys (FK). No previous subdivision exists for Pine Island-Rookery Bay area (PI-RB). In all studies where subdivisions have been reported, a combination of Principal Component and Cluster Analysis was used for grouping FIU's sampling sites, except in the work by Hunt and Todt (2006) where a direct cluster analysis was performed to group the stations. Caccia and Boyer (2005) used FIU WQ data from 1994 to 2003 to subdivide Biscayne Bay into five spatial zones, and Hunt and Todt (2006) combined DERM and FIU data and also found five geographic domains in Biscayne Bay. Boyer et al. (1997) subdivided Florida Bay into four classes and Briceño and Boyer (2010) obtained six water classes. The process by which the bays were subdivided considered several biogeochemical variables and followed an ecological approach, so its results mimic very closely those geographical patterns of: water circulation and residence time (Wang et al. 1994, 2003, 2007; Brand 2001; Cosby et al. 2005;); salinity (Robblee et al. 1989; Boyer et al. 1997; Fourqurean and Robblee 1999; Nuttle et al. 2000; Cosby et al. 2005); TP and TN (Fourqurean et al. 1993; Fourqurean and Robblee 1999; Hitchcock et al. 2007); phytoplankton biovolumes and type (Philips and Badylak 1996; Philips et al. 1999; Steidinger et al. 2001; Hunt and Nuttle 2007); seagrass distribution (Zieman et al., 1989, 1991; Robblee et al. 1991; Fourqurean et al. 2003); and bottom composition (Wanless et al. 1984).

OBJECTIVE

The main objective of this work is to subdivide south Florida's estuaries and coastal waters into basins with similar geomorphological and geochemical characteristics to support the derivation of numeric nutrient criteria.

METHODOLOGY

Data source

Florida International University WQMN (<http://serc.fiu.edu/wqmnetwork>).

Period of Record

The PORs were selected depending upon data availability, variable set completeness, and data quality as shown in Table 1. For BB we reassessed our recent subdivision of Central and South BB (from 09/30/93 to 9/18/08) with a new one that includes North BB and Manatee Bay and Barnes Sound (6/7/96 to 9/24/09), and for FB we adopted the recently published subdivision of Briceño and Boyer (2010) whose POR is from 3/14/91 to 12/07/07. Some of Briceño and Boyer's abbreviations for the basins have been modified to maintain a consistent nomenclature throughout the area of interest.

Water Quality data

FIU WQMN field water column measurements and sampling were performed monthly. Reported measurements included surface and bottom salinity (PSU), temperature (°C), dissolved oxygen (mg/l), and turbidity (NTU). Laboratory determinations under NELAC certified analytical procedures included Total Nitrogen (TN), Total Phosphorus (TP), Total Organic Carbon (TOC), Total Organic Nitrogen (TON), Chlorophyll *a* (CHL_A), Nitrate+Nitrite (NO_x⁻), Nitrite (NO₂⁻),

Nitrate (NO_3^-), Ammonium (NH_4^+), Inorganic Nitrogen (DIN, calculated), Soluble Reactive Phosphate (SRP), and Silicate (SiO_2).

FACTOR ANALYSIS AND CLUSTERING

We re-assessed the subdivisions in the FK, WWB-TTI and BB, and for the first time subdivided the SoFlo (South Florida) west coast estuaries from Pine Island to Rookery Bay (PI-RB). We followed an ecological (multivariate) approach to classify the estuarine and coastal waters. Biogeochemical variables with less than 10% non-detects were selected for Factor Analysis (StatView 5.0.1). Factor Analysis of standardized data used Orthotran/Varimax rotation and PC extraction. Scores were retained and their mean, standard deviation, median, and median absolute deviation at each sampling station were used for hierarchical cluster analysis (SYSTAT 8.0) with Ward distance calculations. A group of small lakes along the Everglades–Florida Bay ecotone (Frankovich et al. 2010) were provisionally incorporated in FB as an additional water type (Coastal Lakes, CL). We are in the process of collecting pertinent water quality data to validate and confirm this grouping. Table 1 summarizes the inputs and results of the whole data processing, indicating the POR, variables used for Factor Analysis, number of stations included, the % variance accounted and the resulting clusters after the Hierarchical cluster analysis.

Table 1

Region	WWB-TTI	FB*	FK	PI-RB	BB1 (Central-South BB)	BB2 (North-Central-South BB)	Shelf
POR	Sep/92-Sep/09	Mar/91 to Dec/07	Mar/95-Oct/09	Jan/99-Sep/09	Sep/93 to Sep/08	Jun/96 to Sep/09	May/95-Sep/07
Input	TN TP	TN TP	TN TP	TN TP	TN TP	TN TP	TP TN
Variables	CHLA TOC	CHLA TOC	CHLA TOC	CHLA TOC	CHLA TOC	CHLA TOC	CHLA NOX
for	SAL DO	SAL DO	SAL DO	SAL_0 DO_0	SAL_0 DO_0	SAL_0 DO_0	NH4 TOC
Factor	TURB NH4	TURB TON	TURB TEMP	TURB NO3	TURB NOX	TURB NOX	SAL_0 DO_0
Analysis		NO3 NO2 NH4 SRP TEMP		NO2 NH4 SRP	NO2 NH4 SRP	NO2 NH4 SRP	TURB
Stations	47	28	155	29	21	30	49
PC	4	6	4	5	5	5	4
Acct Variance	75%	79%	66%	81%	76%	73%	63%
Clusters	8	4 (*2)	5	6	6 (*1)	9	3

*Modified results from Briceño and Boyer (2010) plus CL

Three GIS map layers (shape files) were prepared for this study (Table 2). These layers portray the model boxes as polygons and the FIU station locations as points with the third layer showing the land areas of South Florida. The latter was made by combining the U.S Census Bureau land area maps for each county into a single layer; however, for Miami-Dade County, a shoreline derived from Lidar data and more accurate than the Census version was used.

Table 2: Shape files

Shape File Name	Type	No. of Items	Comment
Modelbox1.shp	Polygon	33	Used to show model boxes.
FIU Wqsites All.shp	Point	335	Used to show FIU station locations.
SouthFla-solid.shp	Polygon	7	Used to show land areas and counties.

Finally, this statistical clustering is a first cut at partitioning ecological regions of the bays and coastal areas according to overall water quality characteristics (Table 3). We plan to further compare pertinent nutrient variables among groups in an effort to determine their statistical significance. Should zones not be significantly different, similar nutrient criteria will be applied to reduce complexity.

TABLE 3

Region	No.	Key	Subbasin
BISCAYNE BAY	1	NNB	Northern North Bay
	2	SNB	Southern North Bay
	3	NCO	North Central Outer Bay
	4	NCI	North Central Inshore
	5	SCO	South Central Outer Bay
	6	SCI	South Central Inshore
	7	SCM	South Central Mid-Bay
	8	CS	Card Sound
	9	MIB	Macatee Bay and Holmes Sound
FLORIDA BAY	1	ECFB	East-Central Florida Bay
	2	NFB	Northern Florida Bay
	3	CL	Coastal Lakes
	4	CFB	Central Florida Bay
	5	SFB	Southern Florida Bay
	6	WFB	Western Florida Bay
FLORIDA KEYS	1	OCEAN	Oceanside
	2	BAY	Bayside
	3	BACK	Backcountry
	4	MAR	Marinas
	5	DRTO	Dry Tortugas
Region	No.	Key	Subbasin
GULF SHELF	1	IBS	Inner Gulf Shelf
	2	MOS	Middle Gulf Shelf
	3	OBS	Outer Gulf Shelf
WHTEWATER BAY TO TEN THOUSAND ISS.	1	WWD	Whitewater Bay
	2	PD	Ponce De Leon
	3	SOM	Shark River mouth
	4	MR	Mangrove River Zone
	5	CTZ	Coastal Transition Zone
	6	DWW	Inner Waterway
	7	CI	Coastal Islands
	8	BLK	Blackwater River
PINE IS. TO ROCKERY BAY	1	MARC	Mario Island
	2	NPL	Naples Bay
	3	CI	Collier Inshore
	4	IBI	Idaho Bay
	5	SCB	San Carlos Bay
	6	PIS	Pine Island Sound

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